CHAPTER 7

CSS MODELING

This chapter discusses the use of modeling to characterize the combined sewer system (CSS) and evaluate CSO control alternatives. It discusses different approaches to identifying the appropriate level of modeling, based on site-specific considerations, and describes the various types of available models. Because of the site-specific nature of CSSs, the varying information needs of municipalities, and the numerous available models, *it does not recommend a specific model or modeling approach.*

7.1 THE CSO CONTROL POLICY AND CSS MODELING

The CSO Control Policy refers to modeling as a tool for characterizing a CSS and the impacts of CSOs on receiving waters. Although not every CSS needs to be analyzed using complex computer models, EPA anticipates that most permittees will need to perform some degree of modeling to support CSO control decisions.

The CSO Control Policy describes the use of modeling as follows:

> *Modeling - Modeling of a sewer system is recognized as a valuable tool for predicting sewer system response to various wet weather events and assessing water quality impacts when evaluating different control strategies and alternatives. EPA supports the proper and effective use of models, where appropriate, in the evaluation of the nine minimum controls and the development of the long-term CSO controlplan. It is also recognized that there are many models which may be used to do this. These models range from simple to complex. Having decided to use a model, the permittee should base its choice of a model on the characteristics of its sewer system, the number and location of overflow points, and the sensitivity of the receiving water body to the CSO discharges... The sophistication of the model should relate to the complexity of the system to be modeled and to the information needs associated with evaluation of CSO control options and water quality impacts.* (Section II.C.1.d)

The Policy also states that:

> *The permittee should adequately characterize through monitoring, modeling, and other means as appropriate, for a range of storm events, the response of its sewer system to wet*

weather events including the number, location and frequency of CSOs, volume, concentration and mass of pollutants discharged and the impacts of the CSOs on the receiving waters and their designated uses. (Section II.C.1)

Finally, the CSO Control Policy also states:

EPA believes that continuous simulation models, using historical rainfall data, may be the best way to model sewer systems, CSOs, and their impacts. Because of the iterative nature of modeling sewer systems, CSOs, and their impacts, monitoring and modeling efforts are complementary and should be coordinated. (Section II.C.1.d)

The CSO Policy supports continuous simulation modeling (use of long-term rainfall records rather than records for individual storms) for several reasons. Long-term continuous rainfall records enable simulations to be based on a sequence of storms so that the additive effect of storms occurring close together can be examined. They also enable storms with a range of characteristics to be included. When a municipality uses the presumption approach, long-term simulations are appropriate because the performance criteria are based on long-term averages, which are not readily determined from design storm simulations. Continuous simulations do not require highly complex models. Models that simulate runoff without complex simulation of sewer hydraulics (e.g., STORM, SWMM RUNOFF) may be appropriate where the basic hydraulics of the system are simple or have been analyzed using a more complex model. In the second case, the results from the more complex model can be used to enable proper characterization of system hydraulics in the simple model.

Running a model in both continuous mode and single event mode can be useful for some systems. When only long-term hourly rainfall data are available, it may be desirable to calibrate the model using more refined single event rainfall data before running the model in continuous mode. For instance, if a CSS is extremely responsive to brief periods of high-intensity rainfall, this may not be adequately depicted using hourly rainfall data.

The CSO Control Policy also states that after instituting the nine minimum controls (NMC), the permittee should assess their effectiveness and should

> *submit any information or data on the degree to which the nine minimum controls achieve compliance with water quality standards (WQS). These data and information should include results made available through monitoring and modeling activities done in conjunction with the development of the long-term CSO control plan described in this Policy.* (Section II.B)
>
> *The purpose of the system characterization, monitoring and modeling program initially is to assist the permittee in developing appropriate measures to implement the nine minimum controls and, if necessary, to support development of the long-term CSO control plan. The monitoring and modeling data also will be used to evaluate the expected effectiveness of both the nine minimum controls, and, if necessary, the long-term CSO controls, to meet WQS.* (Section II.C.1)

The long-term control plan (LTCP) should be based on more detailed knowledge of the CSS and its receiving waters than is necessary to implement the NMC. The LTCP should consider a reasonable range of alternatives, including various levels of controls. Hydraulic modeling may be necessary to predict how a CSS will respond to various control scenarios. A computerized model may be necessary for a complex CSS, especially one with looped networks or sections that surcharge. In simpler systems, however, basic equations (e.g., Hazen-Williams or Manning equation - see Section 5.3.1) and spreadsheet programs can be used to compute hydraulic profiles and predict the hydraulic effects of different control measures. (Verification using monitoring data becomes more important in these latter situations.)

Finally, modeling can support either the presumption or demonstration approaches of the CSO Control Policy. The *demonstration approach* requires demonstration that a proposed LTCP is adequate to meet CWA requirements. Meeting this requirement can necessitate detailed CSS modeling as an input to receiving water impact analyses. On the other hand, the *presumption approach* involves performance-based limits on the number or volumes of CSOs. This approach may require less modeling of receiving water impacts, but is acceptable only if "*thepermitting authority determines that such presumption is reasonable in light of the data and analysis conducted in the characterization, monitoring, and modeling of the system and the consideration of sensitive* *areas* . . . .” (Section II.C.4.a) Therefore, the presumption approach does *not* eliminate the need to consider receiving water impacts.

7.2 MODEL SELECTION STRATEGY

This section discusses how to select a CSS model. Generally, the permittee should use the simplest model that meets the objectives of the modeling effort. Although complex models usually provide greater precision than simpler models, they also require greater expense and effort. This section does not describe all of the available CSS-related models, since other documents provide this information (see Shoemaker et al., 1992; Donigian and Huber, 1991; WPCF, 1989).

CSS modeling involves hydrology, hydraulics, and water quality:

- **Hydrology** is the key factor in determining runoff in CSS drainage basins. Hydrologic modeling is generally done using runoff models to estimate flows influent to the sewer system. These models provide input data for hydraulic modeling of the CSS.
- **CSS hydraulic modeling** predicts the pipe flow characteristics in the CSS. These characteristics include the different flow rate components (sanitary, infiltration, inflow, and runoff), the flow velocity and depth in the interceptors, and the CSO flow rate and duration.
- **CSS water quality modeling** consists of predicting the pollutant characteristics of the combined sewage in the system, particularly at CSO outfalls and at the treatment plant. CSS water quality is measured in terms of bacterial counts and concentrations of important constituents such as BOD, suspended solids, nutrients, and toxic contaminants.

Since hydraulic models are usually used together with a runoff model or have a built-in runoff component, runoff models are discussed as part of hydraulic modeling in the following sections.

Some models include both hydraulic and water quality components, while others are limited to one or the other. Although CSO projects typically involve hydraulic modeling, water quality modeling in the CSS is less common, and a community may decide to rely on CSS water quality monitoring data instead.

Several factors will dictate whether CSS water quality modeling is appropriate. WPCF (1989) concludes that "simulation of quality parameters should only be performed when necessary and only when requisite calibration and verification data are available[...] Another option is to couple modeled hydrologic and hydraulic processes with measured quality data to simulate time series of loads and overflows." Modeling might not be justified in cases where measured CSS water quality variations are difficult to relate to parameters such as land use, rainfall intensity, and pollutant accumulation rates. For these cases, using statistics (such as mean and standard deviation) of CSS water quality parameters measured in the sewer system can be a valid approach. One limitation of this approach, however, is that it cannot account for the implementation of best management practices (BMPs) such as street sweeping or the use of detention basins.

Exhibit 7-1 shows how model selection can be affected by the status of NMC implementation and LTCP development, and by whether the LTCP will be based on the presumption or demonstration approach. To avoid duplication of effort, the permittee should always consider modeling needs that will arise during later stages of LTCP development or implementation.

Nine Minimum Controls (NMC)

In this initial phase of CSO control, hydraulic modeling can be used to estimate existing CSO volume and frequency and the impacts of implementing alternative controls under the NMC. Typically, in this stage of analysis, modeling focuses more on reductions in CSO magnitude, frequency, and duration than on contaminant transport.

Long-Term Control Plan (LTCP)

EPA anticipates that hydraulic modeling will be necessary for most CSSs regardless of whether the community uses the presumption approach or demonstration approach. Both approaches require accurate predictions of the number and volume of CSO events; under the demonstration approach, this information will help determine the amount and timing of pollutant loadings to the receiving water.

Exhibit 7-1. Relevant CSS Hydraulic and Water Quality Modeling for EPA's CSO Control Policy

	CSS Hydraulic Modeling	CSS Water Quality Modeling
Nine Minimum Controls		
Demonstrate implementation of the nine minimum controls	Simple to complex models of duration and peak flows	Limited - Not usually performed
LTCP "Presumption Approach"		
Limit average number of overflow events per year	Long-term continuous simulations (preferred) or design storm simulation	Limited - Not usually performed
Capture at least 85% of wet weather volume per year	Same	Limited - Not usually performed
Eliminate or reduce mass of pollutants equivalent to 85% capture requirement	Same	Use measured concentrations or simplified transport modeling
LTCP "Demonstration Approach"		
Demonstrate that a selected control program . . . is adequate to meet the water quality-based requirements of the CWA	Design storm simulations and/or Long-term continuous simulations	Use measured concentrations or, in limited cases, contaminant transport simulations

Presumption Approach. The presumption approach is likely to require hydraulic modeling to develop accurate predictions of the number and volume of CSOs. Some level of contaminant transport modeling may also be necessary to ensure that the presumption approach will not result in exceedances of water quality criteria in light of available data. In such cases, loading estimates can be developed using measured concentrations or simplified screening methods, coupled with hydraulic modeling.

Demonstration Approach. Under the demonstration approach, the permittee needs to show that the planned controls will provide for attainment of WQS unless WQS cannot be attained as a result of natural background conditions or pollution sources other than CSOs.

Therefore, CSS modeling under the demonstration approach should describe pollutant loadings to the receiving water body. Since water quality modeling in the CSS is directly linked to water quality modeling in the receiving water, the CSS model must generate sufficient data to drive the receiving water model. Further, the resolution needed for the CSS pollutant transport estimates will depend on the time resolution called for in the receiving water model, which is in turn driven by WQS. For pollutants with long response times in the receiving water (such as BOD and nutrients), the appropriate level of loading information is usually the total load introduced by the CSO event. For pollutants with shorter response times (such as bacteria and acutely toxic contaminants), it may be necessary to consider the timing of the pollutant load during the course of the CSO event.

7.2.1 Selecting Hydraulic Models

Hydraulic models used for CSS simulations can be divided into three main categories:

- **Runoff models** based on Soil Conservation Service (SCS) runoff curve numbers,[1] runoff coefficients, or other similar methods for the generation of flow. These models can estimate runoff flows influent to the sewer system and, to a lesser degree, flows at different points in the system. Runoff models do not simulate flow in the CSS, however, and therefore do not predict such parameters as the flow depth, which frequently control the occurrence of CSOs. (The RUNOFF block of EPA's Storm Water Management Model (SWMM) is an example.[2])

- **Models based on the kinematic wave approximation** of the full hydrodynamic equations.[3] These models can predict flow depths, and therefore flow and discharge volumes, in systems that are not subject to surcharging or back-ups (backwater effects).

[1] SCS runoff curves were developed based on field studies measuring runoff amounts from different soil cover combinations. The appropriate runoff curve is determined from antecedent moisture condition and the type of soil. (Viessman et al., 1977)

[2] The SWMM RUNOFF model also has limited capabilities for flow routing in the CSS.

[3] Flow, which is caused by the motion of waves, can be described by the hydraulic routing technique. This technique is based on the simultaneous solution of the fully hydrodynamic equations (the continuity equation and the momentum equation for varying flow). Under certain conditions, these hydrodynamic equations can be simplified to a one-dimensional continuity equation and a uniform flow equation (in place of the full momentum equation). This is referred to as the kinematic wave approximation (discharge is simply a function of depth). (Bedient and Huber, 1992)

These models require the user to input hydrographs from runoff model results. (The TRANSPORT block of SWMM is an example.)

- **Complex, dynamic models** based on the full hydrodynamic equations. They can simulate surcharging, backwater effects, or looped systems, and represent all pertinent processes. These models require the user to input hydrographs from runoff model results. (The EXTRAN block of SWMM is an example.)

Exhibit 7-2 compares the flow routing capabilities of the three SWMM blocks. Section 7.3 discusses available hydraulic models.

The simpler models were developed to support rapid evaluations of CSSs. They require little input data, are relatively easy to use, and require less computer time than complex models. These features, however, are becoming less significant because complex models with user-friendly pre- and post-processors are now widely available. Advances in computer technology render run-time a secondary issue for all but the largest of applications.

Criteria for the selection of a CSS hydraulic model include:

1. **Ability to accurately represent CSS's hydraulic behavior.** The hydraulic model should be selected with the characteristics of the above three model categories in mind. For example, a complex, dynamic model may be appropriate when CSOs are caused by back-ups or surcharging. Since models differ in their ability to account for such factors as conduit cross-section shapes, special structures, pump station controls, tide simulation, and automatic regulators, these features in a CSS may guide the choice of one model over another.

2. **Ability to accurately represent runoff in the CSS drainage basin.** The runoff component of the hydraulic model (or the runoff model, if a separate hydrologic model is used) should adequately estimate runoff flows influent to the sewer system. It should adequately characterize rainfall characteristics as well as hydrologic factors such as watershed size, slope, soil types, and imperviousness.

3. **Extent of monitoring.** Monitoring usually cannot cover an entire CSS, particularly a large CSS. A dynamic model is more reliable for predicting the behavior of unmonitored overflows, since it can simulate all the hydraulic features controlling the overflow, but it often requires extensive resources for its application. In addition,

Exhibit 7-2. Characteristics of RUNOFF, TRANSPORT, and EXTRAN Blocks of the EPA Storm Water Management Model (SWMM)[1]

Characteristics	Blocks		
	RUNOFF	TRANSPORT	EXTRAN
1. Hydraulic simulation method	Nonlinear reservoir, cascade of conduits	Kinematic wave, cascade of conduits	Complete equations, conduit networks
2. Relative computational expense for identical network schematizations	Low	Moderate	High
3. Attenuation of hydrograph peaks	Yes	Yes	Yes
4. Time displacement of hydrograph peaks	Weak	Yes	Yes
5. In-conduit storage	Yes	Yes	Yes
6. Backwater or downstream control effects	No	No[2]	Yes
7. Flow reversal	No	No	Yes
8. Surcharge	Weak	Weak	Yes
9. Pressure flow	No	No	Yes
10. Branching tree network	Yes	Yes	Yes
11. Network with looped connections	No	No	No
12. Number of preprogrammed conduit shapes	3	16	8
13. Alternative hydraulic elements (e.g., pumps, weirs, regulators)	No	Yes	Yes
14. Dry-weather flow and infiltration generation (base flow)	No	Yes	Yes
15. Pollution simulation method	Yes	Yes	No
16. Solids scour-deposition	No	Yes	No
17. User input of hydrographs/ pollutographs[3]	No	Yes	Yes

[1] After Huber and Dickinson, 1988.

[2] Backwater may be simulated as a horizontal water surface behind a storage element.

[3] The RUNOFF block sub-model is primarily intended to calculate surface runoff, but includes the capability to simulate simple channel conveyance of flows. The TRANSPORT and EXTRAN blocks are sewer conveyance models with no runoff components and thus require user input of hydrographs.

most of these models use a complex finite-difference technique to solve for the governing equations. Sound simulation of hydraulic behavior requires that the modeler achieve numeric stability of the solution technique through the selection of appropriate time and space intervals. In some cases, however, estimates of overflow at unmonitored locations can be made based on monitoring in areas with similar geographic features (like slope, degree of imperviousness, or soil conditions), based on V/R ratios[4] and drainage basin characteristics (see Section 5.3.3).

4. **Need for long-term simulations.** Long-term simulations are desirable to predict CSO frequency, volume, and pollutant loadings over certain time periods, like one year. This information can help support the presumption approach. For large systems, long-term simulations using a complex dynamic model often require lengthy computer run times and may be impractical.

5. **Need to assess water quality in CSS.** If CSS water quality simulations are needed, the permittee should consider the model's capability to simulate water quality. To simulate CSS water quality, it is often better to use actual pollutant concentrations from monitoring results together with modeled CSS flows.

6. **Need to assess water quality in receiving waters.** The pollutants of concern and the nature of the receiving water affect the resolution of the CSO data needed for the water quality analyses. For example, bacteria analysis typically requires hourly rather than daily loading data, and the hydraulic model must be capable of providing this resolution.

7. **Ability to assess the effects of control alternatives.** If control alternatives involve assessing downstream back-ups or surcharging and the effects of relieving them, correct simulation may require use of a dynamic model, since other models do not simulate surcharging or back-ups.

8. **Use of the presumption or demonstration approach.** Some permittees using the first presumption approach option-no more than four untreated overflow events per year--can estimate the number of overflow events fairly accurately by calculating the probability of exceeding storage and treatment capacity. Other permittees may need to account for transient flow peaks, which requires accurate flow routing. The other two presumption approach options-percent volume capture and pollutant load capture-generally require some analysis of the timing and peaking of flows, so a hydraulic simulation approach may be needed.

   If a permittee is using the demonstration approach, receiving water monitoring and/or modeling is necessary. The time intervals for pollutant transport in a receiving water model may influence the time intervals for CSS quality modeling.

[4] V/R is the ratio of the overflow volume to the rainfall depth.

This in turn will constrain the time resolution for CSS hydraulic modeling. The permittee should consider the level of time resolution derived when selecting a model.

9. **Ease of use and cost.** As mentioned above, simple models tend to be easier to use than complete dynamic models. Although user-friendly dynamic models now exist, they are generally commercial models that cost more than public domain models and can be used incorrectly by inexperienced users. Another option is to use commercial pre- and post-processors (or shells) designed to facilitate the use of public domain models such as SWMM. They can provide graphically-oriented, menu-driven data entry and extensive results plotting capabilities at a cost lower than that of complete dynamic models.

   Another issue related to ease of use and accuracy is robustness, which is a model's lack of propensity to become unstable. Instabilities are uncontrolled oscillations of the model's results due to the approximations made in the numerical solution of the basic differential equations. Instabilities tend to occur primarily in fully dynamic models, and are caused by many factors, including incomplete sewer information and short conduits. Resolving model instabilities can be time-consuming and requires extensive experience with the model.

7.2.2 Selecting CSS Water Quality Models

CSS water quality models can be divided into the following categories:

- ***Land Use Loading Models*** - These models provide pollutant loadings as a function of the distribution of land uses in the watershed. Generally, these models attribute to each land use a concentration for each water quality parameter, and calculate overall runoff quality as a weighted sum of these concentrations. Pollutant concentrations for the different land uses can be derived from localized data bases or the Nationwide Urban Runoff Program (NURP), a five-year study initiated in 1978 (U.S. EPA, 1983a). Local data are usually preferable to NURP data since local data are generally more recent and site-specific.

- ***Statistical Methods*** - A more sophisticated version of the previous method, statistical methods attempt to formulate a derived frequency distribution for event mean concentrations (EMCs). The EMC is the total mass of a pollutant discharged during an event divided by the total discharge volume. NURP documents discuss the use of statistical methods to characterize CSO quality in detail (Hydroscience, Inc., 1979) and in summary form (U.S. EPA, 1983a).

- ***Build-Up/Washoff Models*** - These models simulate the basic processes that control runoff quality, accounting for such factors as time periods between events, rainfall intensity, and BMPs. They require calibration and are not regularly used due to the expense and difficulty of defining site-specific rates.

Many models do not address the potentially important role of chemical reactions and transformations within the CSS. Calibration may be difficult because pollutant loading into the CSS is often uncertain.

The permittee should consider the following criteria when selecting a CSS water quality model:

1. **Needs of the receiving water quality simulation.** The time scale of the pollutant concentration simulation in the CSS, and the degree of sophistication of the model, depends partly on the needs of the receiving water quality simulation (if used) and, ultimately, on the level of detail required to demonstrate attainment of WQS. If it is only necessary to estimate the average annual loading to the receiving water, then detailed hourly or sub-hourly simulation of combined sewage quality generally will not be necessary. As noted above, in many cases it is appropriate to combine sophisticated hydraulic modeling with approximate CSS water quality modeling.

2. **Ability to assess control and BMP alternatives.** When the control alternatives under assessment include specific BMPs or control technologies, the CSS water quality model should be sophisticated enough to estimate the effects of these alternatives.

3. **Ability to accurately represent significant characteristics of pollutants of concern.** The pollutants involved in CSS quality simulation can be roughly grouped as bacteria, BOD, nutrients, sediments and sediment-associated pollutants, and toxic contaminants. Most water quality models are designed to handle sediments and nutrients, but not all can model additional pollutants. In some cases, this limitation can be circumvented by using a sediment potency factor, which relates the mass of a given pollutant to sediment transport. However, this alternate approach has limited usefulness for CSO concerns since it is generally not appropriate for bacteria and dissolved metals. As noted earlier, another alternate approach is to combine the results of hydrologic and hydraulic modeling of the CSS with bacteria and dissolved metals concentrations from sampling results to estimate pollutant loads.

4. **Capability for pollutant routing.** Another concern is the model's capability for pollutant routing-i.e., its capacity to account for variability in pollutant concentrations during storm events. Most models translate pollutant concentrations from sources and

CSO quantity to pollutant loading without taking separate account of the timing of pollutant delivery due to transport through the CSS. Some basins deliver the highest concentrations of pollutants in the rising limb of the storm flow (the "first flush" effect). If the CSO loading for such systems is modeled using overflow quantity and average concentrations, inaccuracies may result, particularly if the "first flush" is effectively captured by the POTW or storage.

5. **Expense and ease of use.** Sophisticated water quality models can be expensive to calibrate and generally are more difficult to use. If a simpler model is applicable to the situation and can be properly calibrated, it may be sufficient and can be more accurate.

7.3 AVAILABLE MODELS

Exhibit 7-3 summarizes several runoff and hydraulic models and Exhibit 7-4 summarizes several water quality models. These models have been developed by EPA and the Army Corps of Engineers and are available in the public domain. Some of the models in Exhibit 7-3 are runoff models (such as STORM); others have a runoff component but also simulate flow in the CSS (such as SWMM and Auto-Q-ILLUDAS).

An increasing number of high-quality commercial models and pre-/post-processors are also available. Commercial models can be either custom-developed software or enhanced, more user-friendly versions of popular public domain models. In exchange for the cost of a commercial model, users generally receive additional pre- and/or post-processing capabilities and technical support services. Several of the available commercial models are listed in Exhibit 7-5.[5] Commercial pre/post-processors exist for use with some of the public domain models. Pre-processors can help users prepare their input files for a model. Post-processors provide additional capabilities for analyzing and displaying the model output through graphing, mapping, and other techniques. For

[5] The commercial packages have not been reviewed by EPA and they are subject to continued evolution and change, like all commercial software. This listing is provided to assist potential users; it is not meant to endorse any particular model or imply that models not listed are not acceptable. A recent listing of some available models is found in Mao (1992). Recent developments in sewer and runoff models include linking models to geographic information systems (GIS), computer-aided design (CAD) systems, and receiving water models such as WASP.

Exhibit 7-3. CSS Runoff and Hydraulic Models (Public Domain)

Model Name	Characteristics				
	Hydraulic Time Scales	**Hydraulic Simulation Type**	**Assess Control Alternatives**	**Key to Reviews**	**Major References**
EPA Statistical'	Annual, Event	Runoff Coefficient	No	1,2,3	Hydroscience, 1979 Driscoll et al., 1990
The Simple Method	Annual, Event	Runoff Coefficient	No	1	Schueler, 1987
USGS Regression Method	Annual, Event	Regression	No	1,2	Driver & Tasker, 1988
SLAMM	Continuous-Daily	Water Balance	Limited	1	Pitt, 1986
P8-UCM	Continuous-Hourly	Curve Number	Advanced	1	Palmstrom & Walker, 1990
Auto-Q-ILLUDAS	Continuous-Hourly	Water Balance	Limited	1,3	Terstriep et al., 1990
STORM	Continuous-Hourly	Runoff Coeff./ Curve Number	Limited	1,2,3	HEC, 1977
DR3M-QUAL	Continuous-Sub-hourly	Kinematic Wave	Advanced	1,2,3	Alley & Smith, 1982a & 1982b
HSPF	Continuous-Sub-hourly	Kinematic Wave	Moderate'	1,2,3	Johanson et al., 1984
SWMM	Continuous-Sub-hourly	Kinematic & Dynamic Wave	Advanced	1,2,3	Huber & Dickinson, 1988; Roesner et al., 1988

Notes:
1 Reviewed as "FHWA" by Shoemaker et al., 1992.
2 Can be used for assessment of control alternatives, but not designed for that purpose.

Key to Reviews:
1 Shoemaker et al., 1992.
2 Donigian and Huber, 1991.
3 WPCF, 1989.

Some of the public domain models listed above are available from EPA's Center for Exposure Assessment Modeling (CEAM). CEAM can be contacted at:

CEAM
National Exposure Research Laboratory-Ecosystems Research Division
Office of Research and Development
USEPA
960 College Station Road
Athens, GA 30605-2700
Voice: (706) 355-8400
Fax: (706) 355-8302
e-mail: ceam@epamail.epa.gov
CEAM also has an Internet site at http://www.epa.gov/CEAM/

Exhibit 7-4. CSS Water Quality Models (Public Domain)

Model Name	Characteristics: Quality Time Scales	Pollutant Types	Pollutant Routing-Transport Capability	Pollutant Routing - Transformation Capability	BMP Evaluation Capability
EPA Statistical[1]	Annual	S, N, O	no	no	low
The Simple Method	Annual	S, N, O	no	no	low
USGS Regression Method	Annual	S, N O	no	no	no
Watershed	Annual	S, N, O	no	no	medium
GWLF	Continuous - Daily	S, N	low	no	low
SLAMM	Continuous - Daily	S, N, O	medium	no	medium
PB-UCM	Event	N, O	low	no	high
Auto-Q-ILLUDAS	Continuous - Hourly	S, N, O	medium	no	medium
STORM	Continuous - Hourly	S, N, O	no	no	medium
DR3M-QUAL	Continuous - Sub-hourly	S, N, O[2]	high	no	medium
HSPF	Continuous - Sub-hourly	S, N, O	high	high	high
SWMM	Continuous - Sub-hourly	S, N, O[2]	—[3]	low	high

Notes: 1 Reviewed as "FHWA" by Shoemaker et al., 1992.
2 Other constituents can be modeled by assumption of a sediment potency fraction.
3 SWMM received a low rating from Shoemaker et al. for "weak" quality simulations. This rating may not be justified when SWMM's pollutant routing-transport capabilities are compared to those of other models.

Key to Pollutant Type: S - Sediment N - Nutrients O - Other.

Some of the public domain models listed above are available from EPA's Center for Exposure Assessment Modeling (CEAM). CEAM can be contacted at:

CEAM
National Exposure Research Laboratory-Ecosystems Research Division
U.S. EPA Office of Research and Development
960 College Station Road
Athens, GA 30605-2700
Voice: (706) 355-8400 Fax: (706) 355-8302
e-mail: ceam@epamail.epa.gov
CEAM also has an Internet site at http://www.epa.gov/CEAM/

Exhibit 7-5. Selected Commercial CSS Models

Package Name	Type of Hydraulic Simulation	Water Quality Capability	Contact
Hydra/Hydra Graphics	Dynamic	No	PIZER Incorporated 4422 Meridian Avenue N Seattle, Washington 98103 (800) 222-5332 www.pizer.com
Eagle Point Hydrology Series	Dynamic	No	Eagle Point Software 4131 Westmark Drive Dubuque, Iowa 52002-2627 (800) 678-6565 www.eaglepoint.com
Mouse	Dynamic	Yes	Danish Hydraulic Institute Agern Allé 5 DK-2970 Hørrsholm, Denmark 011-45 45 179 100 www.dhi.dk
HydroWorks	Dynamic	Yes	HR Wallingford, Wallingford Software Howbery Park Wallingford Oxfordshire OX10 8BA, UK 01 1-44(0)1491 835381 www.hrwallingford.co.uk
XP-SWMM32	Dynamic	Yes	BOSS International 6612 Mineral Point Rd. Madison, Wisconsin 53705-4200 (800) 488-4775 www.bossintl.com

example, SWMMDuet[6] allows the integration of SWMM and Arc/INFO for database management and GIS analysis.

These exhibits summarize some important technical criteria, and can be used as a preliminary guide. However, to evaluate the use of a specific model in a particular situation the permittee should refer to the more detailed reviews and major references listed in Exhibits 7-3 and 7-4. Both Shoemaker et al. (1992) and Donigian and Huber (1991) provide preliminary evaluations of the functional criteria, including cost and data requirements. The *Water Resources Handbook* (Mays, 1996) discusses both hydraulic and water quality models and compares their attributes.

[6] SWMMDuet is a SWMM/GIS Interface. Further information can be obtained from the Delaware Department of Natural Resources at (302) 739-3451.

7.4 USING A CSS MODEL

7.4.1 Developing the Model

In developing the model, the modeler establishes initial conditions for various model components (such as the level of discretization) and input data parameters (such as percent imperviousness of subcatchments). These elements are then adjusted through model calibration, which is discussed in the next section.

Until recently the modeler had to compromise between the level of detail in a model (temporal and spatial precision), the mode in which it was run (complex vs. simple), and the time period for the simulation (event vs. continuous). As computer technology continues to improve, limitations in computing power are becoming less of a factor in determining the appropriate level of modeling complexity. However, for increased model complexity to lead to greater accuracy, complex models should be used by knowledgeable, qualified modelers who have sufficient supporting data. In some cases, where detail is not required, a simplified model may save time spent filling the data requirements of the model, preparing tiles, and doing the model runs. Shoemaker et al. (1992, Tables 7 to 9) provides a tabular summary of the main input and output data for each of the models presented in Exhibits 7-3 and 7-4.

The level of discretization (i.e., coarse vs. fine scale) determines how precisely the geometry of the CSS and the land characteristics of the drainage basin are described in the model. At a very coarse level of discretization, the CSS is a black box with lumped parameters and the model (e.g., STORM) primarily simulates CSOs. A more complex approach might be to simulate the larger pipes of the CSS, but to lump the characteristics of the smaller portions of the CSS. Another intermediate level of complexity is to simulate the interceptor when it is the limiting component in the CSS for controlling overflows. Much can be learned about system behavior by simulating interceptor hydraulics in response to surface runoff. More complex simulations would include increasing levels of detail about the system.

In determining the appropriate level of discretization, the modeler must ask:

- What is the benefit of a finer level of detail?
- What is the penalty (in accuracy) in not modeling a portion of the system?

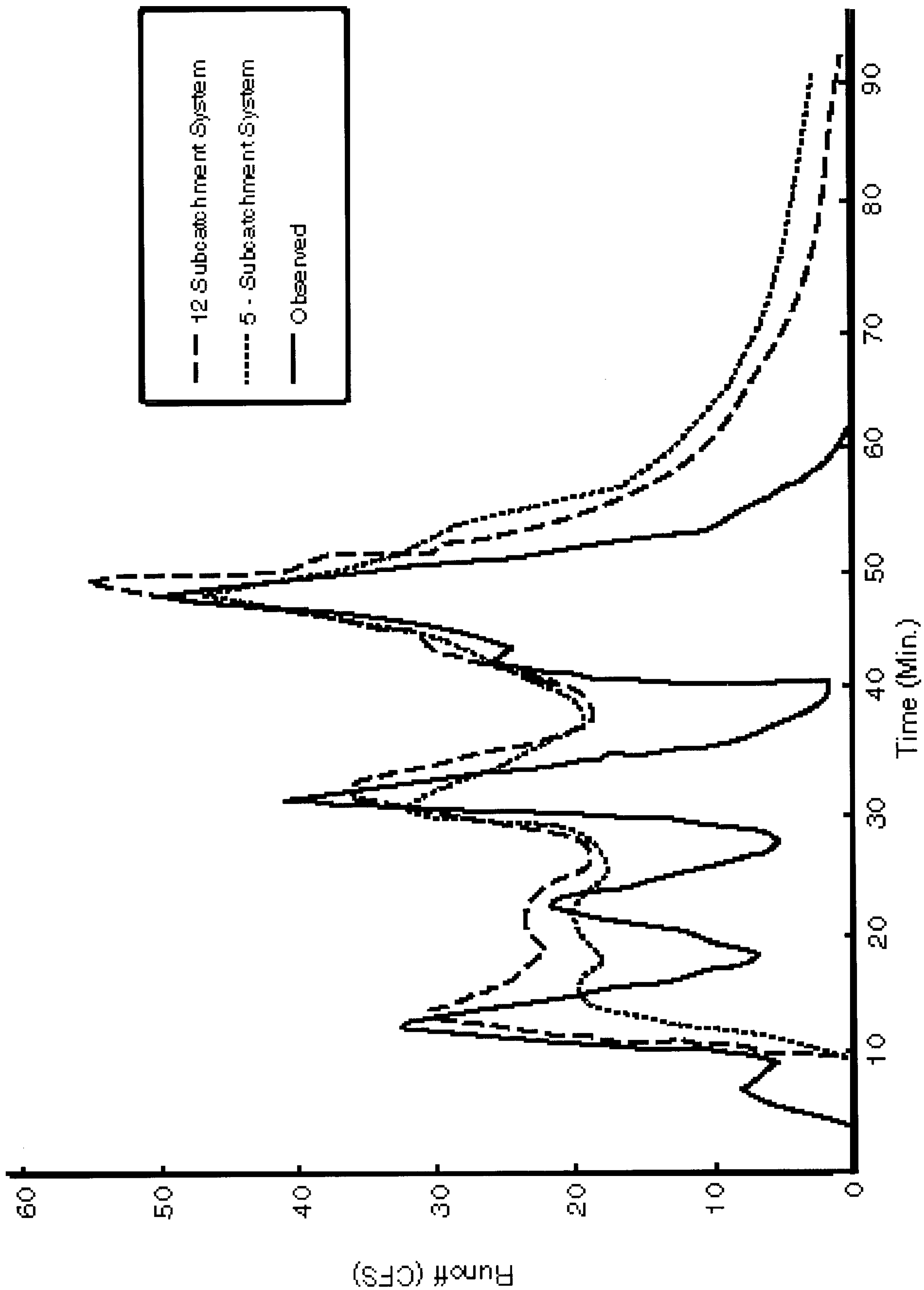
For systems that are controlled hydraulically at their downstream ends, it may only be necessary to model the larger downstream portion of the CSS. If flows are limited due to surcharging in upstream areas, however, a simulation neglecting the upstream portion of the CSS would over-estimate flows in the system. In some cases it is difficult to determine ahead of time what the appropriate level of detail is. In these cases, the modeler can take an incremental approach, determining the value of additional complexity or data added at each step. Exhibit 7-6, for example, compares a simulation based on five subcatchments (coarse discretization) and a simulation based on twelve subcatchments (finer discretization) with observed values. Only marginal improvement is observable when subcatchments are increased from five to twelve. The modeler should probably conclude that even finer discretization (say, 15 subcatchments) would provide little additional value.

7.4.2 Calibrating and Validating the Model

A model general enough to tit a variety of situations typically needs to be adjusted to the characteristics of a particular site and situation. Model calibration and validation are used to “fine-tune” a model to better match the observed conditions and demonstrate the credibility of the simulation results. An uncalibrated model may be acceptable for screening purposes, but without supporting evidence the uncalibrated result may not be accurate. To use model simulation results for evaluating control alternatives, the model must be reliable.

Calibration is the process of running a model using a set of input data and then comparing the results to actual measurements of the system. If the model results do not reasonably approximate actual measurements, the modeler reviews the components of the model to determine if adjustments

Exhibit 7-6. Levels of Discretization



should be made so that the model better reflects the system it represents.[7] For example, a CSS hydraulic model used to simulate overflows is calibrated by running the model using measured rainfall data to simulate the volume, timing, and depth of CSOs. The model results are then compared to actual measurements of the overflows. The modeler then adjusts parameters such as the Manning roughness coefficient or the percent imperviousness of subcatchments within scientifically credible ranges and runs the model a second time, again comparing the results to observations. Initial calibration runs often point to features of the system, such as a connection or bypass, which may not have been evident based on the available maps. The modeler repeats this procedure until satisfied that the model produces reasonable simulations of the overflows. Models are usually calibrated for more than one storm, to ensure appropriate performance for a range of conditions. Exhibit 5-9 shows some example model calibration plots of flow and depth during storm events. For calibration, the most important comparisons are total volumes, peak flows, and shapes of the hydrographs.

Validation is the process of testing the calibrated model using one or more independent data sets. In the case of the hydraulic simulation, the model is run without any further adjustment using independent set(s) of rainfall data. Then the results are compared to the field measurements collected concurrently with these rainfall data. If the results are suitably close, the model is considered to be validated. The modeler can then use the model with other sets of rainfall data or at other outfalls. If validation fails, the modeler must recalibrate the model and validate it again using a third independent data set. If the model fails a validation test, the next test must use a new data set. (Re-using a data set from a previous validation test does not constitute a fair test, because the modeler has already adjusted model parameters to better fit the model to the data.) Validation is important because it assesses whether the model retains its generality; that is, a model that has been adjusted extensively to match a particular storm might lose its ability to predict the effects of other storms.

[7] Model calibration is not simply "curve fitting" to meet the data. Model adjustments should make the modeled elements of the system better reflect the actual system.

The availability of adequate calibration data places constraints on which models are appropriate. When identifying the time period for conducting CSS flow monitoring, the permittee should consider the effect of using larger data sets. The *Combined Sewer Overflow Control Manual* (U.S. EPA, 1993) states that "an adequate number of storm events (usually 5 to 10) should be monitored and used in the calibration." The monitoring period should indeed cover at least that many storms, but calibration and validation are frequently done with 2 to 3 storms each.

EPA's *Compendium of Watershed-Scale Models for TMDL Development* (Shoemaker et al., 1992) includes the following comments on calibration and validation:

> *Most models are more accurate when applied in a relative rather than an absolute manner. Model output data concerning the relative contribution... to overall pollutant loads is more reliable than an absolute prediction of the impacts of one control alternative viewed alone. When examining model output. . . it is important to note three factors that may influence the model output and produce unreasonable data. First, suspect data may result from calibration or verification data that are insufficient or inappropriately applied. Second, any given model, including detailed models, may not represent enough detail to adequately describe existing conditions and generate reliable output. Finally, modelers should remember that all models have limitations and the selected model may not be capable of simulating desired conditions. Model results must therefore be interpreted within the limitations of their testing and their range of application. Inadequate model calibration and verification can result in spurious model results, particularly when used for absolute predictions. Data limitations may require that model results be used only for relative comparisons.*

Common practice employs both judgment and graphical analysis to assess a model's adequacy. However, statistical evaluation can provide a more rigorous and less subjective approach to validation (see Reckhow et al., 1990, for a discussion of statistical evaluation of water quality models).

Nix (1990) suggests the following general sequence for calibrating a CSS model:

1. **Identify the important model algorithms and parameters.** A combination of sensitivity analysis and study of model algorithms can determine which parameters are most important for calibration of a given model-site pairing.

2. **Classify model parameters** to determine the degree to which they can be directly measured, or, alternatively, are conceptual parameters not amenable to direct measurement. For instance, a parameter such as area is usually easily defined, and thus not varied in calibration, while parameters that are both important to model performance and not amenable to direct measurement (e.g., percent imperviousness) will be the primary adjustment factors for calibration.

3. **Calibrate the model first for the representation (prediction) of overflow volume.**

4. After obtaining a reasonable representation of event overflow volume, **calibrate to reproduce the timing and peak flow (hydrograph shape) of overflows.**

5. Finally, **calibrate the pollutant parameters** only after an acceptable flow simulation has been obtained.

Section 7.5 describes an example of CSS modeling, including commentary on calibration and simulation accuracy.

7.4.3 Performing the Modeling Analysis

Once a model has been calibrated and validated, it can be run for long-term simulations and/or for single events (usually a set of design storms).

- **Long-term simulations** can account for the sequencing of the rainfall in the record and the effect of having storms immediately follow each other. They are therefore useful for assessing the long-term performance of the system under the presumption approach. Long-term simulations also assess receiving water quality accurately under the demonstration approach. Water quality criteria need to be evaluated with the frequency and duration of exceedance in order to be relevant. This is best done using long-term continuous simulations or skillfully done probabilistic simulations. Although continuous simulation models should be calibrated using continuous data where possible, they may be calibrated with single events if antecedent conditions are taken into account. As the speed of desktop computers increases, modelers may be able to perform long-term continuous simulations with higher and higher levels of detail.

- **Single event simulations** are useful for developing an understanding of the system (including the causes of CSOs) and formulating control measures, and can be used for calibrating models.

Although increased computer capabilities enable continuous simulations with greater levels of detail, continuous simulation of very large systems can have some drawbacks:

- The model may generate so much data that analysis and interpretation are difficult
- Limitations in the accuracy of hydrologic input data (due to the inability to continuously simulate spatially variable rainfall over a large catchment area) may lead to an inaccurate time series of hydraulic conditions within the interceptor
- The more storms that are simulated, the greater the chance that instabilities will occur in complex models. Correctly identifying and resolving these instabilities requires capable, experienced modelers.

7.4.4 Modeling Results

Model Output

The most basic type of model output is text files in which the model input is repeated and the results are tabulated. These can include flow and depth versus time in selected conduits and junctions, as well as other information, such as which conduits are surcharging. The model output may include an overall system mass balance with such measures as the runoff volume entering the system, the volume leaving the system at the downstream boundaries, the volume lost due to flooding, and the change of volume in storage. The model output can also measure the mass balance accuracy of the model run, which may indicate that problems, such as instabilities (see Section 7.2.1) occurred.

Most models also produce plot tiles, which are easier to evaluate than text files. Output data from plot files can be plotted using spreadsheet software or commercial post-processors, which are available for several public domain models (particularly SWMM). Commercial models typically include extensive post-processing capabilities, allowing the user to plot flow or depth versus time at any point in the system or to plot hydraulic profiles versus time along any set of conduits.

Interpretation of Results

Simulation models predict CSO volumes, pollutant concentrations, and other variables at a resolution that depends on the model structure, model implementation, and the resolution of the input data. Because the ultimate purpose of modeling is generally to assess the CSO controls needed to provide for the attainment of WQS, the model's space and time resolution should match that of the applicable WQS. For instance, a State WQS may include a criterion that a one-hour average concentration not exceed a given concentration more than once every 3 years on average. Spatial averaging may be represented by a concentration averaged over a receiving water mixing zone, or implicitly by the specification of monitoring locations to establish whether the instream criteria can be met. In any case, the permittee should note whether the model predictions use the same averaging scales as the relevant water quality criteria. When used for continuous rather than event simulation, as suggested by the CSO Control Policy, simulation models provide output that can be analyzed to predict the occurrence and frequency of water quality criteria exceedances.

In interpreting model results, the permittee needs to be aware that modeling usually will not provide exact predictions of system performance measures such as overflow volumes or exceedances of water quality criteria. With sufficient effort, the permittee often can obtain a high degree of accuracy in modeling the hydraulic response of a CSS, but results of modeling pollutant buildup/washoff, transport in the CSS, and fate in receiving waters are considerably less accurate. Achieving a high degree of accuracy may be more difficult in a continuous simulation because of the difficulty of specifying continually changing boundary conditions for the model parameters.

In interpreting model results, the permittee should remember the following:

- Model predictions are only as accurate as the user's understanding and knowledge of the system being modeled and the model being used
- Model predictions are no better than the quality of the calibration and validation exercise and the quality of the data used in the exercise
- Model predictions are only estimates of the response of the system to rainfall events.

Model Accuracy and Reliability

Since significant CSO control decisions may be based on model predictions, the permittee must understand the uncertainty (caused by model parameters that cannot be explicitly estimated) and environmental variability (day-to-day variations in explicitly measurable model inputs) associated with the model prediction. For instance, a model for a CSO event of a given volume may predict a coliform count of 350 MPN/100 ml in the overflow, well below the hypothetical water quality criterion of 400 MPN/100 ml. However, the model prediction is not exact, as observation of an event of that volume would readily show. Consequently, additional information specifying how much variability to expect around the "most likely" prediction of 350 is useful. Obviously, the interpretation of this prediction differs, depending on whether the answer is "likely between 340 and 360" or "likely between 200 and 2000."

Evaluating these issues involves the closely related concepts of model accuracy and reliability. **Accuracy** is a measure of the agreement between the model predictions and observations. **Reliability** is a measure of confidence in model predictions for a specific set of conditions and for a specified confidence level. For example, for a simple mean estimation, the accuracy could be measured by the sample standard deviation, while the reliability of the prediction (the sample mean in this case) could be evaluated at the 95 percent confidence level as plus or minus approximately two standard deviations around the mean.

Modeling as part of LTCP development enables the permittee to demonstrate that a given control option is "likely" to result in compliance with the requirements of the CWA and attainment of applicable WQS. During LTCP development, the permittee will justify that a proposed level of control will be adequate to provide for the attainment of WQS. Therefore, the permittee should be prepared to estimate and document the accuracy and reliability of model predictions.

Evaluating model accuracy and reliability is particularly important for the analysis of wet-weather episodic loading, such as CSOs. Such analysis invariably involves estimation of duration (averaging period) and frequency of excursion above a water quality criterion, regardless of whether the criterion is expressed as average monthly and maximum daily values, or as a maximum concentration for a given design stream flow (e.g., 7Q10). Estimating duration and frequency of excursion requires knowledge of model reliability, and the duration and frequency of the storm events serving as a basis for the model.

Available techniques for quantifying uncertainties in modeling studies include sensitivity analysis for continuous simulations, and first-order error analysis and Monte Carlo simulations for non-continuous simulations:

- **Sensitivity analysis** is the simplest and most commonly used technique in water quality modeling (U.S. EPA, 1995g). Sensitivity analysis assesses the impact of the uncertainty of one or more input variables on the simulated output variables.

- **First-order analysis** is used in a manner similar to sensitivity analysis where input variables are assumed to be independent, and the model is assumed to respond linearly to the input variables. In addition to estimating the change of an output variable with respect to an input variable, first-order error analysis also estimates the output variance.

- **Monte Carlo simulation,** a more complex technique, is a numerical procedure where an input variable is defined to have a certain probability density function (pdf). Before each model run, an input variable is randomly selected from each predefined pdf. By combining the results of several model runs, a pdf can be developed for the output variable which is useful in predicting overall model results. The number of model runs is extremely large compared to the number of runs typically done for sensitivity or first-order error analysis. Monte Carlo analysis can be used to define uncertainty (due to uncertain model coefficients) and environmental variability (using historical records to characterize the variability of inputs such as stream flow).

The main input variables for simulating the impact of CSO loadings are properties of the mean rainfall event (storm event depth, duration, intensity, and interval between events), CSO concentrations of specific pollutants, design flow of the receiving water body, and its background concentrations.[8] The output consists of an assessment of the water quality impact in terms of duration and frequency of exceedances of water quality criteria. CSO pollutant concentrations are the main “uncertain” (sensitive) input variables and can be varied over a range of reasonable values to assess their impact on the resulting water quality. Uncertainty analysis can improve management decisions and indicate the need for any additional data collection to refine the estimated loads. For instance, if a small change in CSO pollutant concentrations results in an extremely large variation in the prediction of water quality, it may be appropriate to allocate resources to more accurately estimate the CSO pollutant concentrations used in the model.

7.5 EXAMPLE SWMM MODEL APPLICATION

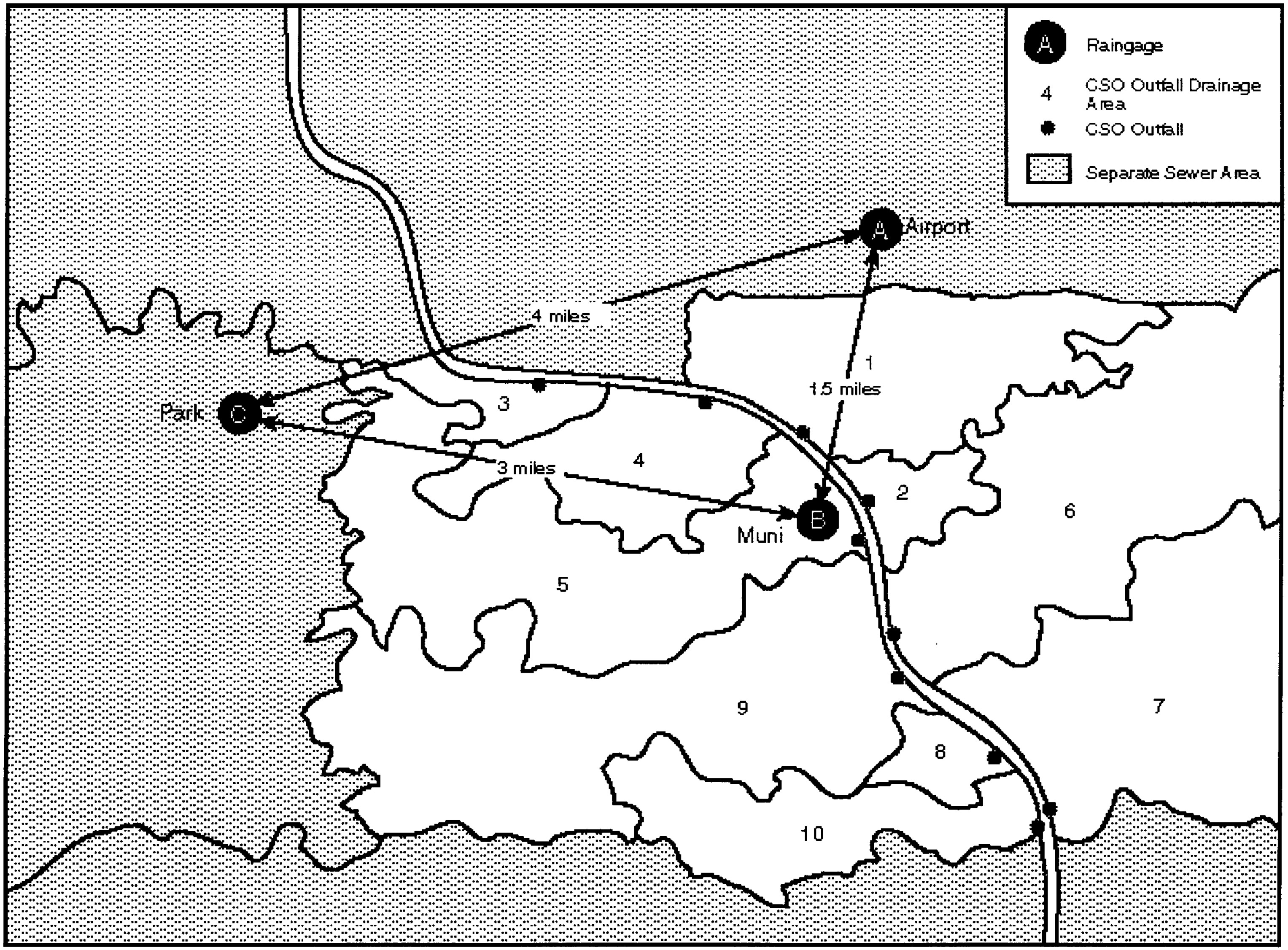
This section applies the Storm Water Management Model (SWMM) to a single drainage area from the example CSS drainage area presented in Chapters 4 and 5. While some of the details of the application are particular to the SWMM model, most of the explanation applies to a range of hydraulic models. The TRANSPORT block of the SWMM model was chosen for the flow routing because the system hydraulics did not include extensive surcharging, and the system engineers felt that a dynamic hydraulic model such as SWMM EXTRAN was not needed to accurately predict the number and volume of CSOs.

7.5.1 Data Requirements

The first step in model application is defining the limits of the combined sewer service area and delineating subareas draining to each outfall (see Exhibit 7-7). This can be done using a sewer system map, a topographic map, and aerial photographs as necessary. The modeler next must decide what portions of the system to model based on their contributions to CSOs (as illustrated in Example 4-1). The modeler then divides selected portions of the CSS and drainage area into segments and translates drainage area and sewer data into model parameters. This process, referred

[8] Continuous simulations do not require use of the “mean” rainfall event or “design” flow data.

Exhibit 7-7. Drainage Area Map



Not to Scale

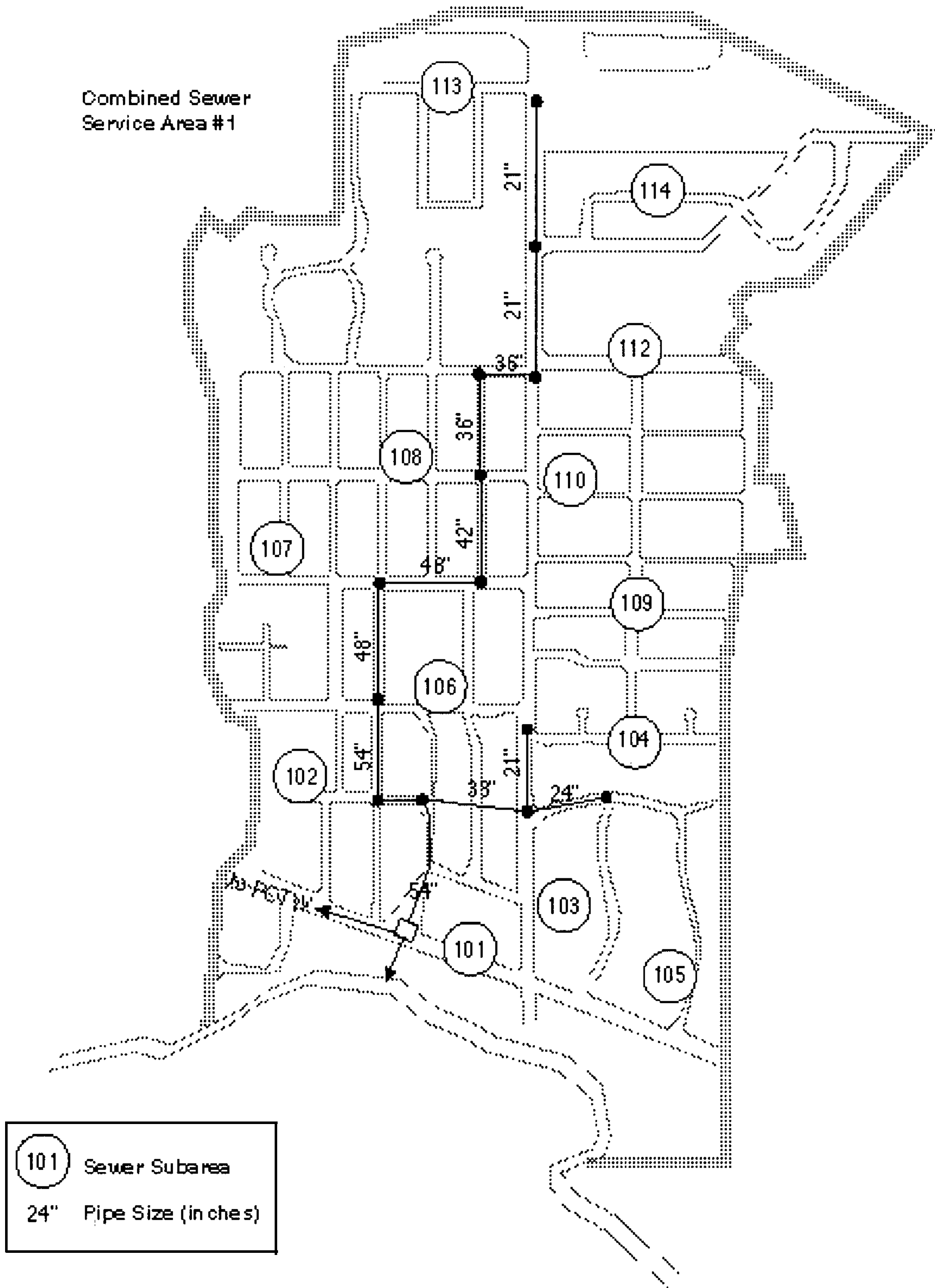
to as discretization, begins with the identification of drainage boundaries, the location of major sewer inlets using sewer maps, and the selection of channels and pipes to be represented in the model. The drainage area is then further divided into subareas, each of which contributes to the nodes of the simulated network.

The modeler must consider the tradeoff between a coarse model that simulates only the largest structures in the CSS, and a fine-scale model that considers nearly every portion of the CSS. A coarse model requires less detailed knowledge of the system, less model development time, and less computer time. The coarse model, however, leaves out details of the system such as small pipes and structures in the upstream end of the CSS. Flow in systems that are limited by upstream structures and flow capacities will not be simulated accurately.

Where pipe capacities limit the amount of flow leaving a drainage area or delivered to the wastewater treatment plant, the modeler should use the flow routing features of the model to simulate channels and pipes in those areas of concern. The level of detail should be consistent with the minimum desired level of flow routing resolution. For example, information cannot be obtained about upstream storage unless the upstream conduits and their subcatchments are simulated. Further, sufficient detail needs to be provided to allow control options within the system to be evaluated for different areas.

In this example, the modeled network is carried to points where the sewers branch into pipes smaller than 21 inches. The system is not directly modeled upstream of these points. Instead, runoff from the upstream area is estimated and routed into the 21-inch pipes. Exhibit 7-8 shows the modeled sewer lines and the subareas tributary to those lines for Service Area 1.

Exhibit 7-8. Sewer Network and Subareas



7.5.2 SWMM Blocks

RUNOFF block. The RUNOFF block of SWMM generates surface runoff and pollutant loads in response to precipitation input and modeled surface pollutant accumulations. The main data inputs for the RUNOFF block are:

- subcatchment width
- subcatchment area
- subcatchment imperviousness
- subcatchment ground slope
- Manning's roughness coefficient for impervious and pervious areas
- impervious and pervious area depression storage
- infiltration parameters.

Exhibit 7-9 shows the main RUNOFF block data inputs (by subcatchment area number) for the example. The subcatchment area is measured directly from maps. Subcatchment width is generally measured from the map, but is more subjective when the subcatchment is not roughly rectangular, symmetrical and uniform. Slopes are taken from topographic maps, and determinations of imperviousness, infiltration parameters, ground slope, Manning's roughness coefficients, and depression storage parameters are based on field observations and aerial photographs.

The RUNOFF block data file is set up to generate an interface file that transfers hydrographs generated by the RUNOFF block to subsequent SWMM blocks for further processing. In this example, the data generated in the RUNOFF block are processed by the TRANSPORT block.

TRANSPORT block. The TRANSPORT block is typically used to route flows and pollutant loads through the sewer system. TRANSPORT also allows for the introduction of dry weather sanitary and infiltration flow to the system. Exhibit 7-10 presents the main TRANSPORT block inputs by element number. It lists the number and type of each element (including upstream elements), the element length (for pipe elements), and inflow (for manholes).

Exhibit 7-9. SWMM Runoff Block Input Parameters (SWMM H1 Card)

						Manning's Coeff.		Depression Storage		Infiltration		
Subarea No.	Inlet No. (manhole)	Width (ft)	Area (ac)	Imperv %	Slope (ft/ft)	Imperv.	Perv.	Imperv.	Perv.	Max Rate (in/hr)	Min Rate (in/hr)	Decay Rate (1/sec)
101	125	3216	25.1	55	.0060	0.015	0.2	0	0.3	1	0.1	0.001
102	126	4114	34.0	35	.0060	0.015	0.2	0	0.3	1	0.1	0.001
103	126	3468	20.7	28	.0125	0.015	0.2	0	0.3	1	0.1	0.001
104	127	4080	28.1	55	.0100	0.015	0.2	0	0.3	1	0.1	0.001
105	128	5140	47.2	22	.0001	0.015	0.2	0	0.3	1	0.1	0.001
106	129	3407	21.9	31	.0040	0.015	0.2	0	0.3	1	0.1	0.001
107	130	7596	27.9	46	.0001	.0150	0.2	0	0.3	1	0.1	0.001
108	130	5614	23.2	38	.0001	0.015	0.2	0	0.3	1	0.1	0.001
109	131	8581	39.4	35	.0170	0.015	0.2	0	0.3	1	0.1	0.001
110	132	5026	20.0	75	.0100	0.015	0.2	0	0.3	1	0.1	0.001
111	133	5445	35.0	17	.0200	0.015	0.2	0	0.3	1	0.1	0.001
112	133	2505	29.9	59	.0140	0.015	0.2	0	0.3	1	0.1	0.001
113	134	7504	37.9	39	.0125	0.015	0.2	0	0.3	1	0.1	0.001
114	135	5610	74.7	29	.0001	0.015	0.2	0	0.3	1	0.1	0.001
115	136	10069	220.0	37	.0100	0.015	0.2	0	0.3	1	0.1	0.001

Exhibit 7-10. SWMM Transport Block Input Parameters (SWMM H1 Card)

Sewer Element Data				Element Type	Inflow (cfs) [for manhole] or Length (ft) [for pipe element]	Pipe Dimension (ft)	Pipe Slope (ft/10 ft)	Manning Pipe Roughness (n)
Element No.	Upstream Element No. 1	Upstream Element No. 2	Upstream Element No. 3					
125	175	0	0	manhole	0.087	NA	NA[1]	NA[1]
175	126	0	0	sewer pipe	1000	.45	0.5	0.014
126	176	177	0	manhole	0.188			
177	150	0	0	sewer pipe	840	2.75	0.28	0.014
150	178	179	0	manhole	0			
178	127	0	0	sewer pipe	390	1.75	0.39	0.014
127	0	0	0	manhole	0.097			
179	128	0	0	sewer pipe	651	2.0	0.34	0.014
128	0	0	0	manhole	0.163			
176	129	0	0	sewer pipe	733	4.5	0.07	0.014
129	180	0	0	manhole	0.076			
180	130	0	0	sewer pipe	841	4.0	0.16	0.014
130	181	0	0	manhole	0.176			
181	131	0	0	sewer pipe	620	4.0	0.09	0.014
131	182	0	0	manhole	0.136			
182	132	0	0	sewer pipe	727	3.5	0.12	0.014
132	183	0	0	manhole	0.103			
183	133	0	0	sewer pipe	771	3	0.16	0.014
133	184	0	0	manhole	0.221			
184	134	0	0	sewer pipe	1110	2.75	0.13	0.014
134	185	0	0	manhole	0.258			
185	135	0	0	sewer pipe	1007	1.75	0.4	0.014
135	0	0	0	manhole	0.131			

[1] Parameter is not applicable for manholes.

The inflow parameter allows for introduction of dry-weather (sanitary) flow to the system. Dry-weather flow is typically distributed proportional to area served. Here it is set to 0.0035 cfs per acre. If the records are available, this parameter can be refined by multiplying the per-capita wastewater flow (typically available from the wastewater treatment plant or latest facilities plan) by the average population density calculated from census figures and sewer service area maps.

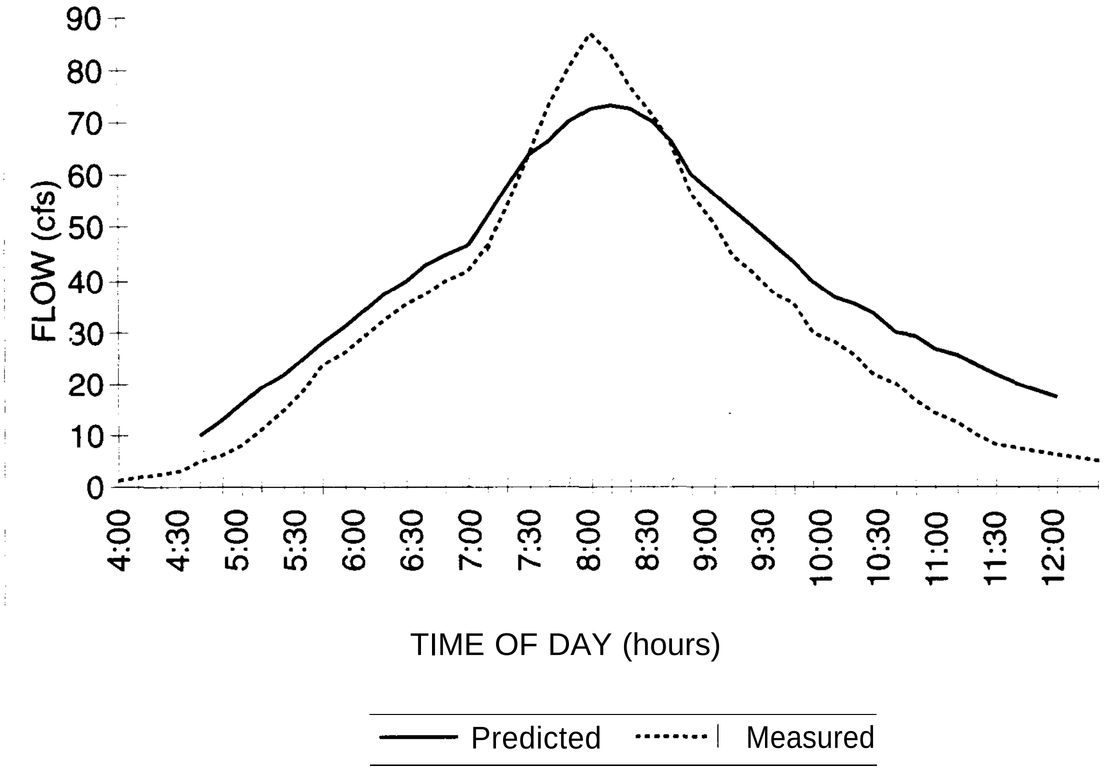
7.5.3 SWMM Hydraulic Modeling

Exhibit 7-11 shows the output hydrograph for element (manhole) 125 from the TRANSPORT block, with the measured flow for the event plotted for comparison. The peak flow, shape of the hydrograph, and the total volume of overflow for thiscalibration run are very close to the measured values.

The SWMM model is applied to monitored drainage areas within the CSS using available monitoring data to calibrate the hydraulic portions of the program to monitored areas. For outfalls that are not monitored, parameters are adjusted based on similar monitored areas and on flow depths or flow determinations obtained from the initial system characterization (see Chapter 3). Once the entire CSS drainage area is modeled and the SWMM model calibrated, the model then needs to be validated. It can then be used to predict the performance of the system for single events (actual or design) and/or for a continuous rainfall record. Recall that it is desirable to calibrate the model to a continuous sequence of storms if is to be applied to a continuous rainfall record. Individual storms related to monitored events can be run to calculate the total volume of overflow for the system. Peak flow values from the SWMM hydrographs can be used for preliminary sizing of conveyance facilities that may be needed to alleviate restrictions.

To predict the number of overflows per year, the calibrated model can be run in a continuous mode and/or for design storm events. In the continuous mode the model can be run using the long-term rainfall record (preferable where the data are available), or for a shorter period of time (e.g., for a typical or extreme year from the example discussed throughout Chapter 5). While the event mode is useful for some design tasks and for estimating hourly loading for a fine-scale receiving water model, the continuous mode is preferable for evaluating the number of overflows under the presumption approach. In this example, the model was run in continuous mode, using data from the

Exhibit 7-11. Flow Hydrograph



38-year rainfall record. The model predicted that between 12 and 32 overflow events would occur per year. The average-22 overflow events per year-is used for comparison with the 4-event-per-year criterion in the presumption approach. (Note that only one outfall in the system needs to overflow to trigger the definition of "CSO event" under the presumption approach.)

Based on model results, system modifications were recommended as part of NMC implementation. After the NMC are in place, the model will be rerun to assess improvement and the need for additional controls.

7.5.4 SWMM Pollutant Modeling

Once the SWMM model has been hydraulically calibrated, it can be used to predict pollutant concentrations in the overflow. The summary of the flow-weighted concentrations generated by the model can then be compared to composite values of actual samples taken during the course of the overflow. Plots of individual concentrations versus time (pollutographs) can also be used to match the variation in concentration of a pollutant during the course of the overflow. First flush effects can also be observed from the model output if buildup/washoff is used.

Model Results

Exhibit 7-12 presents the BOD and total solids output of the SWMM model for the example storm. Note that the modeled concentrations of both pollutants follow a similar pattern throughout the overflow event with little if any first flush concentration predicted in the early part of the overflow. The initial loads assigned within the model for this calibrated example were 70 pounds per acre for BOD and 1,000 pounds per acre for total solids. This model was previously calibrated using monitoring data.

Exhibit 7-13 presents predicted and observed values for BOD and total solids concentrations. The observed concentrations are from analyses of composite samples collected in an automated field sampler for this storm. The modeled values give an approximate, but not precise, estimate of the parameters. While some studies have resulted in closer predictions, this discrepancy between predicted and observed pollutant values is not uncommon.

The modeling in this example could be useful for evaluating the CSS performance against the four-overflow-event-per-year criterion in the presumption approach. It could also be used to evaluate the performance of simple controls.

Exhibit 7-12. Pollutographs

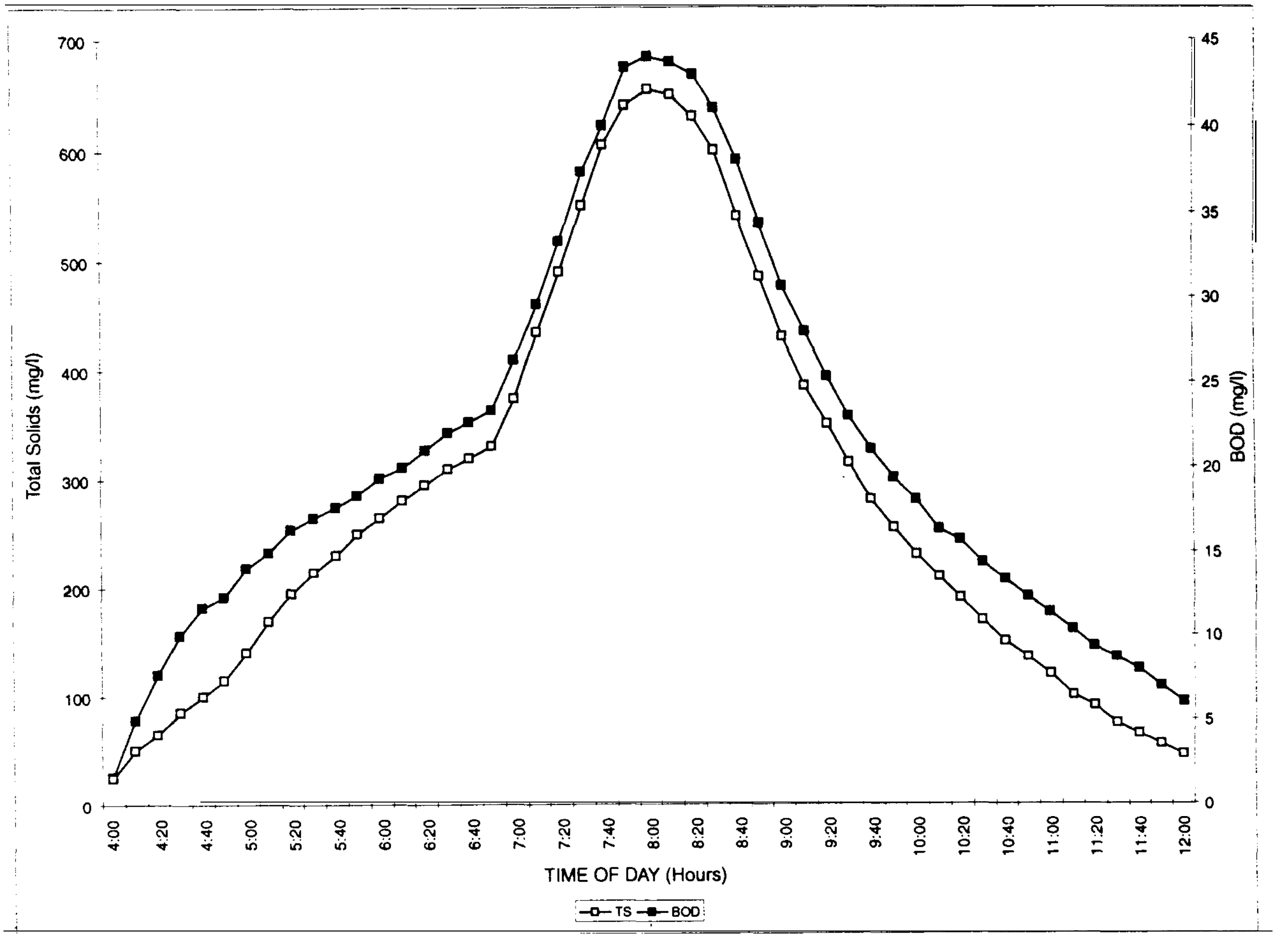


Exhibit 7-13. Predicted and Observed Pollutant Concentrations

	Predicted		Observed	
	BOD	TS	BOD	TS
Flow-weighted concentration (mg/l)	31.4	420	94	300

7.6 CASE STUDY

Example 7-1 is a case study illustrating the CSS and CSO modeling strategy that was developed and implemented by the City of Indianapolis, Indiana. The City, after carefully evaluating available options and regulatory requirements, developed this modeling strategy to characterize system hydraulics and estimate average annual CSO characteristics (i.e., volume, frequency, percent capture, and pollutant loads). The City used the CSS and CSO models to determine CSO impacts on the receiving streams (the White River and its tributaries within the City's combined sewer area), and is now using the models to evaluate various CSO controls and develop an LTCP.

Recognizing that the interceptor sewers and regulators, not the combined sewers, control wet-weather system conveyance capacity to the wastewater treatment plants (and therefore control the occurrences of CSOs), the City used SWMM/EXTRAN to develop a detailed model of interceptor sewers and regulators that included approximately 82 miles of sewer, 173 regulators, and 134 outfalls. The City used SWMM/RUNOFF to generate runoff flows from drainage subcatchments and to calibrate wet-weather flow to the EXTRAN model. The City then used the linked RUNOFF/EXTRAN models to establish critical input data for the STORM model of the CSS, specifically the regulator/interceptor capacities (STORM "treatment rates") and the impervious area estimates (STORM "C" coefficients). The City performed long-term (44-year) continuous simulations using STORM to compute average annual CSO characteristics. The selected modeling strategy enabled the City of Indianapolis to accurately determine interceptor sewer conveyance and system storage capacities, identify system optimization projects, characterize overflows and pollutant loads to receiving streams, and evaluate various CSO control strategies.

Example 7-1. Modeling Case Study — Indianapolis, Indiana

The City of Indianapolis has a population of 741,952 (1990 census) and is the largest city in Indiana. As reported in the City's CSO Operational Plan (December, 1995), the service area includes a combined sewer area of approximately 41 square miles with approximately 82 miles of interceptor sewer, 173 regulators, and 134 outfalls. Pipe sizes in the interceptor sewer system range from 12 inches to 120 inches.

Model Development Strategy

The City used a two-phased modeling strategy to characterize its CSS. Phase 1 focused on the system of interceptor sewers and regulators that deliver flow to the City's advanced WWTPs for treatment, since this part of the system controls the occurrence of CSOs. Phase 1 modeling analysis using SWMM/EXTRAN supported the characterization of the system (required under the CSO Operational Plan), determined the hydraulic capacity of the interceptor sewer system to capture combined sewer flows for treatment, and identified low-cost capital improvement projects to maximize flows to the WWTPs.

For initial analysis and hydraulic characterization of CSOs, the City generated inflow hydrographs to the interceptor model using a ramped hydrography function, which is a synthetic approximation of the rising limb of the actual inflow hydrographs associated with most rain events. The site-specific inflow rates are defined as a function of the ramp slope and the impervious area within the watershed tributary to each model inflow node. Ramped inflow hydrographs are more effective than observed or design event hydrographs for analyzing and evaluating interceptor sewer system capacities and identifying constraints in the system. However, ramp hydrographs cannot be used to calibrate the EXTRAN and STORM models since the response to precipitation must be simulated for calibration. For these reasons, the City used ramped inflow hydrographs in Phase 1 to estimate interceptor system capacities and the SWMM/RUNOFF model for model calibration, and for more detailed analysis, in Phase 2. This let the City efficiently perform initial analysis and hydraulic characterization of CSOs and identify low-cost capital improvement projects to maximize the capture of combined sewer flows in Phase 1, even before model calibration was complete.

In Phase 2, modeling focused on characterization of sewersheds using a more detailed hydrologic (rainfall/runoff) model (RUNOFF) and linking this model directly to the EXTRAN interceptor model developed in Phase 1. The City used the linked models with flow monitoring data from a network of eight flow monitors in the interceptor system and rainfall data for calibration and verification of the interceptor model. Phase 2 modeling also focused on developing and calibrating the CSO model (STORM), using flow monitoring and sampling data at four representative outfalls and simulations to characterize the volume, frequency, and pollutant loads of CSOs. Using regulator/interceptor capacities ("T") from the EXTRAN model, and impervious area estimates ("C") from RUNOFF, the City performed continuous simulations using STORM and the available historical (44-year) hourly precipitation data to generate average annual CSO statistics. STORM can efficiently perform long-term simulations because it uses constant values for "T" and "C", which in the prototype system may vary during long-term simulation. For example, "C" values may vary due to changes in soil moisture conditions in the subcatchments. Therefore a range of values for the CSO characteristics were obtained to reflect these variations in system behavior.

CSS and CSO Characterization Results

As a result of Phase 1 modeling, the City developed its CSO Operational Plan to implement the NMC. The plan used the system conveyance capacity and in-system storage analyses to define a program of hydraulic modifications to the system at 28 individual locations. These modifications enhanced the capture of combined flows during wet weather and reduced overflows to the area's smaller and most sensitive CSO receiving streams. During Phase 2, the City determined average annual CSO characteristics for each CSO outfall, for each major drainage system, and on a system-wide basis using the STORM model of the CSS.

STORM simulations determined that an average annual CSO volume of 4,000 to 5,500 million gallons is discharged from the CSS; CSOs occur at an average frequency of 24 per year; and the interceptor system captures 59 to 66 percent of average annual wet weather combined sewage flow. STORM simulations were also used to estimate that the CSS discharges 1.8 to 2.5 million pounds of BOD and 6.3 to 8.5 million pounds of TSS to the receiving streams on an average annual basis. Based on Phase 2 modeling, the City identified five initial CSO facility projects to demonstrate the effectiveness of various CSO control alternatives. These facilities are now under construction and STORM simulations have been used to estimate that untreated CSO volumes will be reduced by over 80 percent at these five locations.

CHAPTER 8

RECEIVING WATER MODELING

This chapter discusses the use of receiving water modeling to evaluate CSO impacts to receiving waters. It uses the term "modeling" broadly to refer to a range of receiving water simulation techniques. This chapter introduces simplified techniques, such as dilution and decay equations, and more complex computer models, such as QUAL2EU and WASP.

8.1 THE CSO CONTROL POLICY AND RECEIVING WATER MODELING

Under the CSO Control Policy a permittee should develop a long-term control plan (LTCP) that provides for attainment of water quality standards (WQS) using either the demonstration approach or presumption approach. Under the demonstration approach, the permittee documents that the selected CSO control measures will provide for the attainment of WQS, including designated uses in the receiving water. Receiving water modeling may be necessary to characterize the impact of CSOs on receiving water quality and to predict the improvements that would result from different CSO control measures. The presumption approach does not explicitly call for analysis of receiving water impacts.

In many cases, CSOs discharge to receiving waters that are water quality-limited and receive pollutant loadings from other sources, including nonpoint sources and other point sources. The CSO Control Policy states that the permittee should characterize the impacts of the CSOs and other pollution sources on the receiving waters and their designated uses (Section II.C.1). Under the demonstration approach, "[w]here WQS and designated uses are not met in part because of natural background conditions or pollution sources other than CSOs, a total maximum daily load, including a wasteload allocation and a load allocation, or other means should be used to apportion pollutant loads." (Section II.C.4.b)

Established under Section 303(d) of the CWA, the total maximum daily load (TMDL) process assesses point and nonpoint pollution sources that together may contribute to a water body's impairment. This process relies on receiving water models.

An important initial decision-which water quality parameters to model-should be based on data from receiving water monitoring. CSOs affect several receiving water quality parameters. Since the impact on one parameter is frequently much greater than on others, relieving this main impact will likely also relieve the others. For example, if a CSO causes exceedances of bacteria WQS by several hundredfold, as well as moderate dissolved oxygen (DO) depressions, solving the bacterial problem will likely solve the DO problem and so it may be sufficient to monitor bacteria only. Reducing the scope of modeling in this fashion may substantially reduce costs.

8.2 MODEL SELECTION STRATEGY

A receiving water model should be selected according to the following factors:

- The type and physical characteristics of the receiving water body. Rivers, estuaries, coastal areas, and lakes typically require different models.
- The water quality parameters to be modeled. These may include bacteria, DO, suspended solids, toxics, and nutrients. These parameters are affected by different processes (e.g., die-off for bacteria, settling for solids, biodegradation for DO, adsorption for metals) with different time scales (e.g., hours for bacterial die-off, days for biodegradation) and different kinetics. The time scale in turn affects the distance over which the receiving water is modeled (e.g., a few hundred feet for bacteria to a few- miles for DO).
- The number and geographical distribution of CSO outfalls and the need to simulate sources other than CSOs.

This section discusses some important considerations for hydrodynamic and water quality modeling of receiving waters, and how these considerations affect the selection and use of a model.

The purpose of receiving water modeling is primarily to predict receiving water quality under different CSO pollutant loadings and flow conditions in the receiving water. The flow conditions, or hydrodynamics, of the receiving water are an important factor in determining the effects of CSOs on receiving water quality. For simple cases, hydrodynamic conditions can be determined from the receiving water monitoring program; elsewhere a hydrodynamic model may be necessary.

Hydrodynamic and water quality models are either ***steady-state*** or ***transient.*** Steady-state models assume that conditions do not change over time, while transient models can simulate conditions that vary over time. Flexibility exists in the choice of model types; generally, either a steady-state or transient water quality simulation can be done regardless of whether flow conditions are steady-state or transient.

8.2.1 Hydrodynamic Models

A hydrodynamic model provides the flow conditions, characterized by the water depth and velocity, for which receiving water quality must be predicted. The following factors should be considered for different water body types:

- ***Rivers-*** Rivers generally flow in one direction (except for localized eddies or other flow features) and the stream velocity and depth are a function of the flow rate. The flow rate in relatively large rivers may not increase significantly due to wet weather discharges, and a constant flow can be used as a first approximation. This constant flow can be a specified low flow, the flow observed during model calibration surveys, or a flow typical of a season or month. When the increase of river flow is important, it can be estimated by adding together all upstream flow inputs or by doing a transient flow simulation. The degree of refinement required also depends on the time scale of the water quality parameters of interest. For example, assuming a constant river flow may suffice for bioaccumulative toxicants (e.g., pesticides) because long-term exposure is ofimportance. For DO, however, the time variations in river flow rate may be need to be considered.

- ***Estuaries-*** CSO impacts in estuaries are affected by tidal variations of velocity and depth (including reversal of current direction) and by possible salinity stratification. Tidal fluctuations can be assessed by measuring velocity and depth variations over a tide cycle or by using a one- or two-dimensional model. Toxics with relatively small mixing zones can be analyzed using steady currents corresponding to different times during the tidal cycle, but this may require using a computed circulation pattern from a model.

- ***Coastal Areas-*** CSO impacts in coastal areas are also affected by tidal fluctuations. The discussion on estuaries generally applies to coastal areas, but, because the areas are not channelized, two-dimensional or even three-dimensional models may be necessary.

- ***Lakes-*** CSO impacts in lakes are affected by wind and thermal stratification. Wind-driven currents can be monitored directly or simulated using a hydrodynamic model (which may need to cover the entire lake to simulate wind-driven currents properly). Thermal stratification can generally be measured directly.

Because the same basic hydrodynamic equations apply,[1] some of the major models for receiving waters can be used to simulate more than one type of receiving water body. Ultimately, three factors dictate whether a model can be used for a particular hydraulic regime. One factor is whether it provides a one-, two-, or three-dimensional simulation. A second is its ability to handle specific boundary conditions, such as tidal boundaries.

A third factor is whether the model assumes steady-state conditions or allows for time-varying pollutant loading. In general, models that assume steady-state conditions cannot accurately model CSO problems that require analysis of far-field effects. However, in some instances a steady-load model can estimate the maximum potential effect, particularly in systems where the transport of constituents is dominated by the main flow of the water body, rather than local velocity gradients. For example, by assuming a constant source and following the peak discharge plug of water downstream, the steady-load model QUAL2EU can determine the maximum downstream effects of conventional pollutants. The result is a compromise that approximates the expected impact but neglects the moderating effects of longitudinal dispersion. However, QUAL2EU cannot give an accurate estimate of the duration of excursions above WQS.

8.2.2 Receiving Water Quality Models

The frequency and duration of CSOs are important determinants of receiving water impacts and need to be considered in determining appropriate time scales for modeling. CSO loads are

[1] The basic hydrodynamic equations are for momentum and continuity. The momentum equation describes the motion of the receiving water, while the continuity equation is a flow balance relationship (i.e., total inflows to the receiving water less total outflows is equal to the change in receiving water volume).

typically delivered in pulses during storm events. Selection of appropriate time scales for modeling receiving water impacts resulting from a pulsed CSO loading depends upon the time and space scales necessary to evaluate the WQS. If analysis requires determining the concentration of a toxic at the edge of a relatively small mixing zone, a steady-state mixing zone model may be satisfactory. When using a steady-state mixing zone model in this way, the modeler should apply appropriately conservative but characteristic assumptions about instream flows during CSO events. For pollutants such as oxygen demand, which can have impacts lasting several days and extending several miles downstream of the discharge point, it may be warranted to incorporate the pulsed nature of the loading. Assuming a constant loading is much simpler (and less costly) to model; however, it is conservative (i.e., leads to impacts larger than expected). For pollutants such as nutrients where the response time of the receiving water body may be slow, simulating only the average loading rate, usually over a period of days (e.g., 21 days) depending on the nutrient, may suffice.

Receiving water models vary from simple estimations to complex software packages. The choice of model should reflect site conditions. If the pulsed load and receiving water characteristics are adequately represented, simple estimations may be appropriate for the analysis of CSO impacts. To demonstrate compliance with the CWA, the permittee may not need to know precisely where in the receiving water excursions above WQS will occur. Rather, the permittee needs to know the maximum pollutant concentrations and the likelihood that excursions above the WQS can occur at any point within the water body. However, since CSOs to sensitive areas are given a higher priority under the CSO Policy, simulation models for receiving waters with sensitive areas may need to use short time scales (e.g., hourly pollutant loads), and have high resolution (e.g., several hundred yards or less) to specifically assess impacts to sensitive areas.

8.3 AVAILABLE MODELS

Receiving water models cover a wide variety of physical and chemical situations and, like combined sewer system (CSS) models, vary in complexity. EPA has produced guidance on receiving water modeling as part of the Waste Load Allocation (WLA) guidance series. These models, however, tend to concentrate on continuous sources and thus may not be the most suitable for CSOs. Ambrose et al. (1988a) summarizes EPA-supported models, including receiving water models.

This guidance does not provide a complete catalogue of available receiving water models. Rather, it describes simplified techniques and provides a brief overview of relevant receiving water models supported by EPA or other government agencies. In many cases, detailed receiving water simulation may not be necessary. Use of dilution and mixing zone calculations or simulation with simple spreadsheet models may be sufficient to assess the magnitude of potential impacts or evaluate the relative merits of various control options.

Types of Simulation

Water quality parameters can be simulated using either single-event, steady-state modeling or continuous, dynamic modeling. Many systems may find it beneficial to use both types of modeling.

Many of the simpler approaches to receiving water evaluation assume steady flow and steady or gradually varying loading. These assumptions may be appropriate if an order-of-magnitude estimate or an upper bound of the impacts is required. The latter is obtained by using conservative parameters such as peak loading and low current speed. If WQS attainment is predicted under realistic worst-case assumptions, more complex simulations may not be needed.

Due to the random nature of CSOs, the use of dynamic simulation may be preferable to single-event, worst-case, steady-state modeling. Dynamic techniques allow the modeler to derive the fraction of time during which a concentration was exceeded and water quality was impaired. For instance, when using daily simulated results, specific concentrations are first ranked with the corresponding number of occurrences during the simulation period. Frequency distribution plots are then developed and used to determine how often the l-day-acute water quality criteria are likely to be exceeded. The same approach can be used to develop frequency distributions for longer periods such as 4-day or 30-day average concentrations. EPA (1991a) recommends three dynamic modeling techniques: continuous simulation, Monte Carlo simulation, and lognormal probability modeling.

Continuous simulation models solve time-dependent differential equations to simulate flow volume and water quality in receiving waters. These deterministic models incorporate the manner in which flow and toxic pollutant concentrations change over time in a continuous manner rather than relying on simplified terms for rates of change. They use daily effluent flow and concentration data with daily receiving water flow and concentration data to estimate downstream receiving water concentrations. If properly calibrated and verified, a continuous simulation model can predict variable flow and water quality accurately-although at a considerable time and resource expenditure, however.

Monte Carlo simulation is generally used for complex systems that have random components. Input variables are sampled at random from pre-determined probability distributions and used in a toxic fate and transport model. The distribution of output variables from repeated simulations is analyzed statistically to derive a frequency distribution. However, unlike continuous simulation models, the temporal frequency distribution of the output depends on the temporal frequency distribution of the input data. For instance, if the water quality criterion is based on a 4-day average, the input variables must use the probability distributions based on a 4-day average.

Lognormal probability modeling estimates the same output variable probability distributions as continuous and Monte Carlo simulations but with less effort. However, like Monte Carlo simulation, the input must be probability distributions based on input data for the specific temporal frequency distribution desired. The theoretical basis of the technique permits the stochastic nature of the CSO process to be explicitly considered. This method assumes that each of the four variables that affect downstream receiving water quality (rainfall, runoff, event mean concentration of contaminant in the runoff (EMC), and streamflow) can be adequately represented by a lognormal probability distribution. When the EMC is coupled with a lognormal distribution of runoff volume, the distribution of runoff loads can be derived. The storm water load frequency is then coupled with a lognormal distribution of streamflow to derive the probability distribution of in-stream concentrations. The main advantage of lognormal probability modeling is that the probability distributions can be derived using only the median and the coefficient of variation for each input variable.

8.3.1 Model Types

The following sections discuss techniques for simulating different water quality parameters in rivers, lakes and estuaries.

RIVERS

Bacteria and Toxics. Bacteria and toxic contaminants are primarily a concern in the immediate vicinity of CSO outfalls. They are controlled by lateral mixing, advection, and decay processes such as die-off (for bacteria), vaporization (for toxics), and settling and resuspension (for bacteria and toxics). When stream flow is small relative to CSO flow, lateral mixing may occur rapidly and a one-dimensional model may be appropriate. Initial estimates can be made using a steady-state approach that neglects the time-varying nature of the CSO. In this case, concentrations downstream of a CSO are given by:

$$C_x = \frac{Q_u C_u + Q_e C_e}{Q_s} e^{\frac{-KX}{u}}$$

where:[2]

C_x	=	max pollutant concentration at distance X from the outfall (M/L^3)
C_e	=	pollutant concentration in effluent (M/L^3)
C_u	=	pollutant concentration upstream from discharge (M/L^3)
Q_e	=	effluent flow (L^3/T)
Q_u	=	stream flow upstream of discharge (L^3/T)
Q_s	=	stream flow downstream of discharge, $Q_u + Q_e$ (L^3/T)
X	=	distance from outfall (L)
u	=	stream flow velocity (L/T)
K	=	net decay rate (die-off rate for bacteria, settling velocity divided by stream depth for settling, resuspension velocity divided by stream depth for resuspension, vaporization rate divided by stream depth for vaporization) (1/T)
e	=	2.71828...

Since bacteria and toxics can settle out of the water column and attach to sediments, sediments can contain significant amounts of these pollutants. Resuspension of sediments and subsequent desorption of bacteria and toxics into the water column can be an important source of receiving water contaminants. Modeling of sediment resuspension requires estimation of

[2]M=unit of mass, L=unit of length, and T=unit of time.

resuspension velocities and knowledge of sediment transport processes. Thomann and Mueller (1987) discusses how to determine the solids balance in a river and estimate sediment resuspension velocities. Modeling of sediment transport is complex and is often done using computer models such as WASP5 and HSPF.

In large rivers, lateral mixing may occur over large distances and bacterial counts or toxics concentrations on the same shore as the discharge can be calculated using the following expression, as a conservative estimate (U.S. EPA, 1991a):

$$C_x \approx \frac{C_e Q_e W}{Q_s \sqrt{\dfrac{\pi D_y X}{u}}}$$

where: D_y = lateral dispersion coefficient (L^2/T)
W = stream width (L)
π = 3.14159...

This equation is conservative because it neglects any discharge-induced mixing. Simulating over the correlated probability distributions of C_e, Q_e, Q_s, and Q_u can provide an estimate of the frequency of WQS exceedances at a specific distance from the outfall. The method requires the estimation of a lateral dispersion coefficient, which can be measured in dye studies or by methods described in Mixing ***in Inland and Coastal Waters*** (Fischer et al., 1979). Fischer's methods calculate the lateral dispersion coefficient D_y as follows:

$$D_y = 0.6\ du^* \pm 50\%$$

where: ***d*** = water depth at the specified flow (L)
u^* = shear velocity (L/T).

In turn, the following equation estimates shear velocity:

$$u^* = (gds)^{1/2}$$

where: g = acceleration due to gravity (L/T^2)
s = slope of channel (L/L)
d = water depth (L).

The model DYNTOX (LimnoTech, 1985) is specially designed for analysis of toxics in rivers and can handle all three dynamic modeling techniques. U.S. EPA (1991a) and the WLA series by Delos et al. (1984) address the transport of toxics and heavy metals in rivers.

Oxygen Demand/Dissolved Oxygen. The time scales and distances affecting DO processes are greater than for bacteria and toxics. Lateral mixing therefore results in approximately uniform conditions over the river cross section and one-dimensional models are usually appropriate for simulation. The WLA guidance (U.S. EPA, 1995g) discusses the effects of steady and dynamic DO loads, and provides guidelines for modeling impacts of steady-state sources. Simple spreadsheet models such as STREAMDO IV (Zander and Love, 1990) have recently become available for DO analysis.

In general, screening analyses using classical steady-state equations can examine DO impacts to rivers as a result of episodic loads. This approach assumes plug flow, which in turn allows an assumption of constant loading averaged over the volume of the plug (Freedman and Marr, 1990). This approach does not consider longitudinal diffusion from the plug, making it a conservative approach. The plug flow analysis should correlate with the duration of the CSO. For example, a plug flow simulation of a 2-hour CSO event would result in a downstream DO sag that would also last for 2 hours. Given the plug flow assumption, the classic Streeter-Phelps equation can estimate the DO concentration downstream:

$$D = D_o e^{-K_a t} + \frac{W}{Q}\left(\frac{K_d}{K_a - K_r}\right)\left[e^{-K_r t} - e^{-K_a t}\right]$$

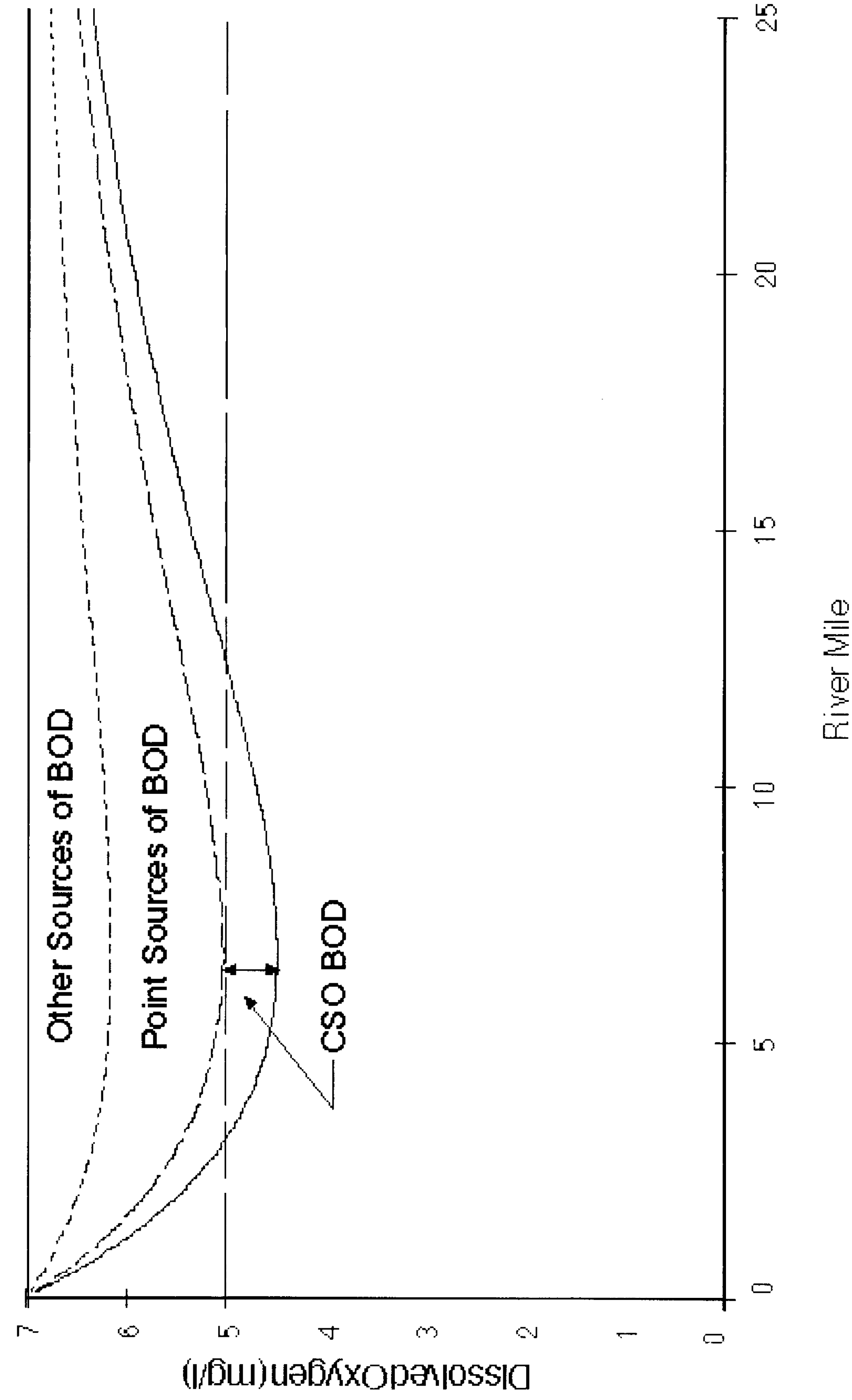
where:
D = DO deficit downstream (M/V)
D_0 = initial DO deficit (M/V)
K_a = atmospheric re-aeration rate (1/T)
t = time of passage from source to downstream location (T)
W = total pollutant loading rate (M/T)
Q = total river flow (V/T)
K_d = biochemical oxygen demand (BOD) deoxygenation rate (1/T)
K_r = BOD loss rate (1/T).

This method can address the joint effects of multiple steady sources through the technique of superposition (Exhibit 8-1). Superposition is used when linear differential equations, such as the Streeter-Phelps equation, govern pollutant concentrations along a receiving stream. For such linear systems, the concentration of a pollutant in a river due to multiple steady-state sources is the linear summation of the responses due to the individual sources. Superposition techniques are also used to estimate pollutant concentrations due to multiple steady-state sources of toxic pollutants. However, it cannot address multiple sources that change over time, nor can it address the effects of river morphology. When such issues are important, more sophisticated modeling techniques are necessary.

More sophisticated modeling techniques are also necessary to assess the effects of sediment oxygen demand (SOD) and plant respiration (which remove oxygen from the receiving water), and photosynthesis by aquatic plants (which adds oxygen to the water). The Streeter-Phelps equation makes the simplifying assumption that there are only point sources of CBOD, so SOD, photosynthesis, and respiration are assumed to be zero. If photosynthesis, respiration, and SOD are significant, more complex analysis is needed to evaluate these factors. These distributed sources and sinks of DO and BOD are addressed by Thomann and Mueller (1987) and by several computer models, including QUAL2EU and WASPS.

Nutrients/Eutrophication. Nutrient discharges affect river eutrophication over time scales of several days to several weeks. Nutrient/eutrophication analysis considers the relationship between

Exhibit 8-1. Dissolved Oxygen Superposition Analysis



nutrients and algal growth. Analysis of nutrient impacts in rivers is complex because nutrients and planktonic algae,[3] which are free-floating one-celled algae, usually move through the system rapidly.

The current WLA guidance (U.S. EPA, 1995g) considers only planktonic algae (rather than all aquatic plants) and discusses nutrient loadings and eutrophication in rivers primarily as a component in computing DO. The guidance applies to narrative criteria that limit nuisance plant growth in large, slowly flowing rivers.

LAKES

Bacteria and Toxics. Mixing zone analysis can often be used to assess attainment of WQS for bacteria and toxics in lakes. For a small lake in which the effluent mixes rapidly, the concentration response is given by the following equation (Freedman and Marr, 1990):

$$C = \frac{M}{V} e^{(-K - \frac{Q}{V})t}$$

where: C = concentration (M/L^3)
M = mass loading (M)
Q = flow (L^3/T)
K = net decay rate (bacteria die-off, settling and resuspension, volatilization, photolysis, and other chemical reactions) (1/T)
V = lake volume (L^3)
t = time (T).

For an incompletely-mixed lake, however, a complex simulation model is generally necessary to estimate transient impacts from slug loads. The EPA WLA guidance series contains a manual on chemical models for lakes and impoundments (Hydroqual, Inc., 1986). This guidance, which also applies to bacteria, describes simple and complex models and presents criteria for selecting models and model parameters.

[3] Aquatic plants can be divided into those that move freely with the water (planktonic aquatic plants) and those that are attached or rooted in place.

Oxygen Demand/Dissolved Oxygen. Simple analytical approximations can model oxygen demand and DO in cases where DO mixing occurs quickly relative to depletion by COD/BOD. Where lateral mixing occurs rapidly but vertical temperature stratification exists, DO concentration can be addressed for a two-layer stratified lake under the following simplifying assumptions (from Thomann and Mueller, 1987):

- The horizontal area is constant with depth
- Inflow occurs only to the surface layer
- Photosynthesis occurs only in the surface layer
- Respiration occurs at the same rate throughout the lake
- The lake is at steady-state.

With these severe restrictions, the solution is given by:

$$c_1 = \left(\frac{q}{K_L + q}\right)c_o + \left(\frac{K_L}{K_L + q}\right)c_s + \frac{pH_1 - RH - S_B}{K_L + q} - \frac{K_{d1}H_1L_1 - K_{d2}H_2L_2}{K_L + q}$$

and

$$c_2 = c_1 - \left(\frac{S_B + RH_2 - K_{d2}H_2L_2}{E/H_i}\right)$$

where the subscripts 1 and 2 refer to the epilimnion (top layer) and hypolimnion (lower layer), respectively, and variables without subscripts refer to the whole lake, and where:

q	=	Outflow rate (L/T)
K_L	=	DO transfer rate at lake surface (L/T)
c	=	DO concentration (M/L^3)
c_0, c_s	=	Initial and saturation dissolved oxygen concentrations (M/L^3)
p	=	Gross photosynthetic production of DO (m/L^3-T)
H	=	Depth (L)
H_i	=	$H/2$ when $H_1 = H_2$ and H_1 when $H_2 >> H_1$ (L)
R	=	Phytoplankton DO respiration (M/L^3-T)

S_B = Sediment oxygen demand (M/L^2-T)
K_d = Deoxygenation coefficient (1/T)
L = Steady-state CBOD concentration in water column (M/L^3), = $W/(Q+K_rV)$, where W is the mass loading rate, Q is the rate of flow through the lake, V is the volume, and K_r is the net loss rate.
E = Dispersion coefficient (L^2/T).

Because this analysis assumes steady-state loading and because measuring some of the parameters proves difficult, the method may only have limited application to CSOs. A modeler able to define all of the above parameters may choose to apply a more spatially resolved model.

In many cases, complex simulation models are necessary to analyze DO in lakes. These are either specialized lake models or flexible models, such as EUTROWASP, that are designed to address issues specific to lakes. Some experienced modelers have been successful in modeling thermally stratified lakes with one or two dimensional river models (e.g., QUAL2EU) that assume the river bottom is the thermocline.[4]

Nutrient/Eutrophication Impacts. For lakes, simple analytic equations often can analyze end-of-pipe impacts and whole-lake impacts, but evaluating mixing phenomena frequently requires a complex computer model (Freedman and Marr, 1990). Simple analytical methods can be applied to lake nutrient/eutrophication impacts in situations where the CSOs mix across the lake area within the time scale required to obtain a significant response in the algal population. In most lakes, phosphorus is considered to be the limiting nutrient for nuisance algal impacts and eutrophication. Mancini et al. (1983) and Thomann and Mueller (1987) have developed a procedure for calculating the allowable surface loading rate. The following steps are drawn from this procedure:

Step 1. Estimate the lake volume, surface area, and mean depth.

Step 2. Estimate the mean annual inflow and outflow rates. Where urban areas draining to the lake constitute a significant fraction of the total drainage area, flow

[4] Such techniques should not be used by inexperienced modelers as they can lead to inaccuracies if they are not used with caution.

estimates from urban runoff and CSOs should be included in the hydrologic balance around the lake. For lakes with large surface areas, the estimate should include surface precipitation and evaporation.

Step 3. Determine the average annual total phosphorus loading due to all sources, including all tributary inflows, municipal and industrial sources, distributed urban and rural runoff, and atmospheric inputs. ***Technical Guidance Manual for Performing Waste Load Allocation*** (Mancini et al., 1983) discusses techniques for estimating these loadings.

Step 4. For total phosphorus, assign a net sedimentation loss rate that is consistent with a local data base.

Step 5. Select trophic state objectives of either total phosphorus or chlorophyll-a consistent with local experience. Calculate the value of the allowable phosphorus area1 loading, W^1, from:

$$W' = a\bar{z}\left(\frac{Q}{V} + v_s\right)$$

where: W' is the allowable area1 surface loading rate (M/L^2-T)
a is the trophic state objective concentration of total phosphorus or chlorophyll-a (M/L^3),
Q is outflow (L^3/T),
V is lake volume (L^3),
$\bar{z}$ is mean depth (L), and
v_s is the net sedimentation velocity (L/T).

Step 6. Compare the total area1 loading determined in Step 3 to the value of W^1 obtained in Step 5.

Additional approaches are discussed in Reckhow and Chapra (1983b).

ESTUARIES

Unlike most rivers, estuaries are tidal (i.e., water moves upstream during portions of the tidal cycle and downstream during other parts of the cycle). When averaged on the basis of tidal cycles, pollutant transport in narrow, vertically mixed estuaries with dominant longitudinal flow is similar to that in rivers. However, due to tidal reversals of flow, a narrow estuary may have a much larger effective dispersion coefficient since shifting tides may cause greater lateral dispersion. In such a system, the modeler can apply approximate or screening models used for rivers, provided that an appropriate tidal dispersion coefficient has been calculated. In wider estuaries, tides and winds often result in complex flow patterns and river-based models would be inappropriate. WLA guidance for estuaries is provided in several EPA manuals (Ambrose et al., 1990; Martin et al., 1990; Jirka, 1992; Freedman et al., 1992).

In addition to their tidal component, many estuaries are characterized by salinity-based stratification. Stratified estuaries have the horizontal mixing due to advection and dispersion that is associated with rivers and the vertical stratification characteristic of lakes.

In complex estuaries, accurate analysis of far-field CSO impacts-such as nutrients/ eutrophication, DO, and impacts on particular sensitive areas-typically requires complex simulation models. Simpler analyses are sometimes possible by treating the averaged effects of tidal and wind-induced circulation and mixing as temporally constant parameters. This approach may require extensive site-specific calibration.

Near-field mixing zone analysis in estuaries also presents special problems, because of the role of buoyancy differences in mixing. Jirka (1992) discusses mixing-zone modeling for estuaries.

8.3.2 Computer Models Supported by EPA or Other Government Agencies

This section describes some computer models relevant to receiving water modeling. Most of these models are supported by EPA's Center for Exposure Assessment Modeling (CEAM). CEAM maintains a distribution center for water quality models and related data bases.[5] CEAM-supported models relevant to modeling impacts on receiving water include QUAL2EU, WASPS, HSPF, EXAMSII, CORMIX, MINTEQ, and SMPTOX3. The applicability and key characteristics of the CEAM-supported models are summarized in Exhibit 8-2.

[5] See Section 7.3 for information on obtaining models from CEAM.

Exhibit 8-2. EPA CEAM-Supported Receiving Water Models

Applicability to Hydraulic Regimes and Pollutant Type

	Rivers & Streams			Lakes & Impoundments			Estuaries			Near Field Mixing
Model	**Nutrients**	**Oxygen**	**Other**	**Nutrients**	**Oxygen**	**Other**	**Nutrients**	**Oxygen**	**Other**	
QUAL2EU	✓	✓	✓							
WASP5	✓	✓	✓	✓	✓	✓	✓	✓	✓	
HSPF	✓	✓	✓	✓	✓	✓				
EXAMSII			✓			✓			✓	
CORMIX	Near-field mixing model for all water body types									✓
MINTEQ	Equilibrium metal speciation model									
SMPTOX3			✓							

Key Characteristics and References

Model	Pollutant Loading Type	Transport Dimensionality	Current Version	Key References
QUAL2EU	Steady	1-D	3.22	Brown & Barnwell, 1987
WASP5	Dynamic	Quasi-2/3-D (link-node)	5.10	Ambrose, et al., 1988
HSPF	Dynamic (integrated)	1-D	10.11	Johanson, et al., 1984
EXAMSII	Dynamic	User input (quasi 3-D)	2.96	Burns, et al., 1982
CORMIX	Steady (near field) [1]	Quasi-3-D (zonal)	2.10	Doneker & Jirka, 1990
MINTEQ	Steady	None	3.11	Brown & Allison, 1987
SMPTOX3	Steady	1-D	2.01	LimnoTech, 1992

[1] CORMIX was originally developed assuming steady ambient conditions; Version 3 allows for application to some unsteady environments (e.g., tidal reversal conditions) where transient recirculation and pollutant build-up can occur (CEAM, 1998).

QUAL2EU is a one-dimensional model for rivers. It assumes steady-state flow and loading but allows simulation of diurnal variations in temperature or algal photosynthesis and respiration. QUAL2EU simulates temperature, bacteria, BOD, DO, ammonia, nitrate, nitrite, organic nitrogen, phosphate, organic phosphorus, algae, and additional conservative substances.[6] Because it assumes steady flow and pollutant loading, its applicability to CSOs is limited. QUAL2EU can, however, use steady loading rates to generate worst-case projections for CSOs to rivers. The model has pre- and post-processors for performing uncertainty and sensitivity analyses.

Additionally, in certain cases, experienced users may be able to use the model to simulate non-steady pollutant loadings under steady flow conditions by establishing certain initial conditions or by dynamically varying climatic conditions. If used in this way, QUAL2EU should be considered a screening tool since the model was not designed to simulate dynamic quality conditions.

WASP5 is a quasi-two-dimensional or quasi-three-dimensional water quality model for rivers, estuaries, and many lakes. It has a link-node formulation, which simulates storage at the nodes and transport along the links. The links represent a one-dimensional solution of the advection dispersion equation, although quasi-two-dimensional or quasi-three-dimensional simulations are possible if nodes are connected to multiple links. The model also simulates limited sediment processes. It includes the time-varying processes of advection, dispersion, point and nonpoint mass loading, and boundary exchanges. WASP5 can be used in two modes: EUTRO5 for nutrient and eutrophication analysis and TOXI5 for analysis of toxic pollutants and metals.

WASP5 is essentially a pollutant fate and transport model. Transport can be driven by another hydrodynamic model such as DYNHYD5. DYNHYD5 is a one-dimensional/quasi-two-dimensional model that simulates transient hydrodynamics (including tidal estuaries).

[6] A conservative substance is one that does not undergo any chemical or biological transformation or degradation in a given ecosystem. (U.S. EPA, 1995g)

HSPF is a one-dimensional, comprehensive hydrologic and water quality simulation package which can simulate both receiving waters and runoff to CSSs for conventional and toxic organic pollutants. HSPF simulates the transport and fate of pollutants in rivers and reservoirs. It simulates three sediment types: sand, silt, and clay.

EXAMSII can rapidly evaluate the fate, transport, and exposure concentrations of steady discharges of synthetic organic chemicals to aquatic systems. A recent upgrade of the model considers seasonal variations in transport and time-varying chemical loadings, making it quasi-dynamic. The user must specify transport fields to the model.

CORMIX[7] is an expert system for mixing zone analysis. It can simulate submerged or surface, buoyant or non-buoyant discharges into stratified or unstratified receiving waters, with emphasis on the geometry and dilution characteristics of the initial mixing zone. The model uses a zone approach, in which a flow classification scheme determines which near-field mixing processes to calculate. The CORMIX model cannot be calibrated in the classic sense since rates are fixed based on the built-in logic of the expert system.

MINTEQ determines geochemical equilibrium for priority pollutant metals. Not a transport model, MINTEQ provides a means for modeling metal partitioning in discharges. It provides only steady-state predictions. The model usually must be run in connection with another fate and transport model, such as those described above. A number of assumptions (e.g., equilibrium conditions at the point of mixing between a CSO and the receiving water) must be made to link MINTEQ predictions to another fate and transport model, so it should be used cautiously in evaluating wet weather impacts.

SMPTOX3 is a one-dimensional steady-state model for simulating the transport of contaminants in the water column and bed sediments in streams and non-tidal rivers. SMPTOX3 is an interactive computer program that uses an EPA technique for calculating concentrations of

[7] In some applications CORMIX has proven inaccurate for single port discharges.

toxic substances in the water column and stream bed as a result of point source discharges to streams and rivers. The model predicts pollutant concentrations in dissolved and particulate phases for the water column and bed sediments, as well as total suspended solids. SMPTOX3 can be run at three different levels of complexity: as described above (highest complexity), to calculate toxic water column concentrations but no interactions with bed sediments (medium complexity), or as a total pollutant toxics model (low complexity) (LimnoTech, 1992).

The following additional models are supported by EPA or other government agencies:[8]

DYNTOX is a one-dimensional, probabilistic toxicity dilution model for transport in rivers. It provides continuous, Monte Carlo, or lognormal probability simulations that can be used to analyze the frequency and duration of ambient toxic concentrations resulting from a waste discharge. The model considers dilution and net first-order loss, but not sorption and benthic exchange. DYNTOX Version 2.1 and the draft manual are available from the Office of Science and Technology in EPA's Office of Water (202-260-7012).

CE-QUAL-W2 is a reservoir and narrow estuary hydrodynamics and water quality model developed by the Waterways Experiment Station of the U.S. Army Corps of Engineers. The model provides dynamic two-dimensional (longitudinal and vertical) simulations. It accounts for density effects on flow as a function of the water temperature, salinity and suspended solids concentration. CE-QUAL-W2 can simulate up to 21 water quality parameters in addition to temperature, including one passive tracer (e.g., dye), total dissolved solids, coliform bacteria, inorganic suspended solids, algal/nutrient/DO dynamics (11 parameters), alkalinity, pH and carbonate species (4 parameters).

[8] McKeon and Segna (1987), Ambrose et al. (1988a) and Hinson and Basta (1982) have reviewed some of these models.

8.4 USING A RECEIVING WATER MODEL

As was the case for CSS models (see Section 7.4), receiving water modeling involves developing the model, calibrating and validating the model, performing the simulation, and interpreting the results.

8.4.1 Developing the Model

The input data needs for a specific receiving water model depend upon the hydraulic regime and model used. The permittee should refer to the model's documentation, the relevant sections of the WLA guidance, or to texts such as *Principles of Surface Water Quality Modeling and Control* (Thomann and Mueller, 1987). Tables B-2 through B-5 in Appendix B contain general tables of data inputs.

8.4.2 Calibrating and Validating the Model

Like CSS models, receiving water models need to be calibrated and validated. The model should be run to simulate events for which receiving water hydraulic and quality monitoring were actually conducted, and the model results should be compared to the measurements. Generally, receiving water models are calibrated and validated first for receiving water hydraulics and then for water quality. Achieving a high degree of accuracy in calibration can be difficult because:

- Pollutant loading inputs typically are estimates rather than precisely known values.
- Three-dimensional receiving water models are still not commonly used for CSO projects, so receiving water models involve spatial averaging (over the depth, width or cross-section). Thus, model results are not directly comparable with measurements, unless the measurements also have sufficient spacial resolution to allow comparable averaging.
- Loadings from non-CSO sources, such as storm water, upstream boundaries, point sources, and atmospheric deposition, often are not accurately known.
- Receiving water hydrodynamics are affected by numerous factors which are difficult to account for. Those include fluctuating winds, large-scale eddies, and density effects.

Although these factors make model calibration challenging, they also underscore the need for calibration to ensure that the model reasonably reflects receiving water data.

8.4.3 Performing the Modeling Analysis

Receiving water modeling can involve single events or long-term simulations. Single event simulations are usually favored when using complex models, which require more input data and take significantly longer to run (although advances in computer technology keep pushing the limits of what can practically be achieved.) Long-term simulations can predict water quality impacts on an annual basis.

Although a general goal is to predict the number of water quality criteria exceedances, models can evaluate exceedances using different measures, such as hours of exceedance at beaches or other critical points, acre-hours of exceedance, and mile-hours of exceedance along a shore. These provide a more refined measure of the water quality impacts of CSOs and of the expected effectiveness of different control measures.

CSO loadings commonly are simulated separately from other loadings in order to assess the relative impacts of CSOs. This is appropriate because the equations that best approximate receiving water quality are usually linear and so effects are additive (one exception, however, is the non-linear algal growth response to nutrient loadings).

8.4.4 Using Modeling Results

By calculating averages over space and time, simulation models predict CSO volumes, pollutant concentrations, and other variables of interest. The extent of this averaging depends on the model structure, how the model is applied, and the resolution of the input data. The model's space and time resolution should match that of the necessary analysis. For instance, the applicable WQS may be expressed as a 1-hour average concentration not to exceed a given concentration more than once every three years on average. Spatial averaging may be represented by a concentration averaged over a receiving water mixing zone, or implicitly by the specification of monitoring locations to establish compliance with instream criteria. In any case, the permittee should note whether the model predictions use the same averaging scales required in the permit or relevant WQS.

When used for continuous rather than event simulation, as suggested by the CSO Control Policy, simulation models can predict the frequency of exceedances of water quality criteria. Probabilistic models, such as the Monte Carlo simulation, also can make such predictions. In probabilistic models, the simulation is made over the probability distribution of precipitation and other forcing functions such as temperature, point sources, and flow. In either case, modelers can analyze the output for the frequency of water quality criteria exceedances.

The key result of receiving water modeling is the prediction of future conditions due to implementation of CSO control alternatives. In most cases, CSO control decisions will have to be supported by model predictions of the pollutant load reductions necessary to achieve WQS. In the receiving waters, critical or design water quality conditions might be periods of low flows and high temperature that are established based on a review of available data. Flow, temperature, and other variables for these periods then form the basis for analysis of future conditions.

It is useful to assess the sensitivity of model results to variations in parameters, rate constants, and coefficients. A sensitivity analysis can determine which parameters, rate constants, and coefficients merit particular attention in evaluating CSO control alternatives. The modeling approach should accurately represent features that are fully understood, and sensitivity analysis should be used to evaluate the significance of factors that are not as clearly defined. (See Section 7.4.4 for additional discussion of sensitivity analysis.)